Composting of organic waste: quantification and assessment of greenhouse gas emissions

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DTU Environment
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Preface

The work presented in this PhD thesis, entitled 'Composting of organic waste: quantification and assessment of greenhouse gas emissions', was conducted at the Department of Environmental Engineering at the Technical University of Denmark with Associate Professor Charlotte Scheutz as supervisor and Professor Thomas Højlund Christensen as co-supervisor. The PhD project was conducted from September 2007 to October 2010 and was funded by the Technical University of Denmark.

The content of the PhD thesis is based on eight papers prepared for scientific journals. These papers are referred to in the text, by the names of the authors and their appendix number written with roman numbers.

I Andersen, J.K., Boldrin, A., Samuelsson, J., Christensen, T.H., Scheutz, C., 2010. Quantification of GHG emissions from windrow composting of garden waste. Journal of Environmental Quality. 39:713-724.

II Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Mass balances and life cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Waste Management & Research. DOI:10.1177/0734242X09360216.

III Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Greenhouse gas emissions from home composting of organic household waste. Waste Management. DOI:10.1016/j.wasman.2010.07.004

IV Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Mass balances and life cycle inventory of home composting of organic household waste. Submitted to Waste Management.

V Boldrin, A., Andersen, J.K., Møller, J., Favoino, E., Christensen, T.H., 2009. Composting and compost utilization: Accounting of greenhouse gases and global warming contributions. Waste Management & Research. 27:800-812.

VI Andersen, J.K., Christensen, T.H., Scheutz, C., 2010. Substitution of peat, fertiliser and manure by compost used in hobby gardening: User surveys and cases. Waste Management. DOI:10.1016/j.wasman.2010. 07.011

VII Boldrin, A., Andersen, J.K., Christensen, T.H., 2010. Environmental assessment of garden waste management in the Municipality of Aarhus, Denmark. Submitted to Waste Management.

VIII Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Home composting as an alternative treatment option for organic household waste: an environmental comparison using life cycle assessment-modelling. Submitted to Waste Management.

The papers Andersen et al. (I and II) and Boldrin et al. (V and VII) were also included in the PhD thesis by Alessio Boldrin and Section 4.1 of this thesis was written based on the work presented in Boldrin (2009).

The papers are not included in this www-version, but can be obtained from the Library at DTU Environment
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In addition, the following publications have been produced during the PhD study:

Boldrin, A., Andersen, J.K., Christensen, T.H. 2009. LCA-report: Environmental assessment of garden waste management in Aarhus Kommune. Department of Environmental Engineering, Technical University of Denmark, 2800 Kgs. Lyngby, Denmark.

Møller, J., Andersen, J.K., Christensen, T.H. 2010. Miljøvurdering af udbringning af have-park overskud på landbrugsjord (Environmental assessment of direct application of garden waste on agricultural soil, in Danish). Department of Environmental Engineering, Technical University of Denmark, 2800 Kgs. Lyngby, Denmark.

October 2010
Jacob Kragh Andersen

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My appreciations go to my office-mates and to the entire solid waste research group which have been a great part of my life at DTU Environment, as good colleagues and great friends. Thanks to all present and former colleagues for great inputs, rewarding discussions and (sometimes much needed) social breaks.

Special thanks go to Hans Mosbæk, Karina Henriksen and Sinh Nguyen for their valuable help in the laboratory, Bent Skov for assistance in the field and Torben Dolin who prepared all the graphs and illustrations for the thesis. Also special thanks goes to the staff (especially Henrik Halgaard and Preben Stjernholm) at the composting plant in Aarhus, Denmark for being very helpful during sampling campaigns and for making the working atmosphere very pleasant.

Last, but not least I would like to thank family and friends (you know who you are) who supported me during the entire PhD project and told me that everything was going to be alright.

Jacob

Summary

An increased focus on source separation and the collection of the organic fraction of municipal solid waste (MSW) has led to an increased flow of organic waste to treatment in e.g. composting facilities. Organic waste in the municipal waste stream consists generally of garden waste and organic household waste (OHW) (mainly food waste). Almost all garden waste collected in Denmark is currently treated in windrow composting plants, whereas most OHW is not source-separated and is instead incinerated in waste-to-energy (WtE) plants.

The main focus of the thesis was on contributions (loads and savings) to global warming from composting of organic waste. Data collection and environmental assessments were performed for central composting of garden waste (at a full-scale windrow composting plant in Aarhus, Denmark) and home composting of OHW (in six differently operated home composting units representing composting in single families). The thesis provides emission factors (EFs) (in kg substance Mg^{-1} wet waste (ww)) for the greenhouse gases (GHGs) emitted during composting. These EFs are based on the most comprehensive measuring campaigns performed to date for central and home composting of organic waste. GHG emissions were reported as 2.4 kg CH_4 Mg^{-1} ww and 0.06 kg N_2O Mg^{-1} ww, giving a total EF of 111 kg CO_2-equivalents (eq.) Mg^{-1} ww from central composting of garden waste. EFs from central composting were measured using a dynamic plume measurement method, which combines a controlled tracer gas release with concentration measurements in the downwind plume. It was concluded that pore gas concentration measurements and small-scale emissions measurement methods (such as static flux chambers) were not suitable for measuring GHG emissions from windrow composting of garden waste.

GHG emissions were also assessed from single-family home composting of OHW, for which static flux chambers were used. In this case, the flux chambers were found more suitable since they cover the entire home composting unit. Six home composting units (difference in mixing frequency) were monitored during one year of composting. The emissions were reported as 0.4-4.2 kg CH_4 Mg^{-1} ww and 0.30-0.55 kg N_2O Mg^{-1} ww, with the highest emissions coming from the frequently mixed units and the lowest emissions from the units that were not turned at all. These numbers add up to 100-239 kg CO_2-eq. Mg^{-1} ww. The unit that represented the most realistic management of home composting (low mixing

frequency and low additions of waste) had an EF of 127 kg CO_2 Mg^{-1} ww. This is at the same level as reported from central composting of garden waste; however, it should be noted that the input waste was of different types.

Another focus of the thesis was on the environmental savings associated with the use of compost in gardens and/or horticulture. Compost can be a substitute for inorganic fertiliser and peat in growth media when applied to soil. The substitution rate is a crucial parameter when environmental assessments of composting are performed. The actual substitution, however, is not very well known. User surveys were performed in order to estimate the actual substitution of fertilisers and peat by compost use in private gardens. The reported substitution estimates were 41% and 58% for the Aarhus (n=74) and Copenhagen area (n=1832), respectively. These results indicate that the substitution of fertiliser, peat and manure is far from 100%, which is sometimes assumed in environmental assessments of composting.

GHG EFs and the estimated substitution, together with other collected data (e.g. electricity and fuel consumption) for central composting of garden waste and home composting of OHW were gathered in life cycle inventories (LCIs). These LCIs were used as platforms for performing life cycle assessments (LCAs) of the two mentioned systems. These LCAs were performed by means of the waste-LCA model EASEWASTE, which accounts for all relevant environmental aspects of organic waste treatment across a time horizon of 100 years. The LCAs provided a holistic view of waste treatment and identified the most important and sensible processes contributing to environmental loads during composting.

The LCA of garden waste management in Aarhus showed that the environmental loads from composting were low. The most important processes were identified as GHG emissions (environmental impact) and the substitution of fertilisers and peat in growth media (environmental saving). The assessments showed that additional benefits could be obtained by directing parts of garden waste (woody parts with high lower heating value (LHV) and low ash content) to incineration in WtE plants. Not all garden waste should be incinerated, and it was recommended to maintain the composting plant so there will be compost for use as a substitute for fertilisers and peat. There is a potential for optimizing the environmental profile by increasing the use of compost as a substitute for fertilisers and peat as well as optimizing the composting process in order to decrease the direct GHG emissions. The environmental impacts from home composting of OHW were also quite low,

and important processes were identified as GHG emissions and the potential substitution of fertiliser and peat by compost used in gardens. The assessment showed that in several potential impact categories, home composting performed as well as or better than incineration and landfilling of OHW. One exception was for global warming (GW), in which incineration performed better due to the substitution of fossil fuel (coal-based electricity and heat production).

In general, it was shown that environmental loads from composting of organic waste were low. The LCA results indicate that home composting should be recommended as a supplementary treatment option for parts of the OHW. Apart from the benefits included in the LCAs of composting, a range of benefits exist that are difficult to quantify, but should be taken into account when comparing composting with alternative treatment options. Citizens are quite happy with picking up compost and in general they think that the soil takes on a nicer colour. Savings in nutrients (especially phosphorous), better soil structure and better water holding capacity are just some of the additional benefits from composting of organic waste.

Sammenfatning

Fokus på kildesortering og indsamling af den organiske del af husholdningsaffald har givet en øget mængde af organisk affald til behandling i for eksempel komposteringsanlæg. Organisk affald fra husholdninger består primært af haveaffald og madaffald (kaldet organisk husholdningsaffald (OHA) i denne rapport). Stort set alt det haveaffald, som indsamles i Danmark bliver behandlet i milekomposteringsanlæg, hvorimod det meste af OHA ikke bliver kildesorteret - men i stedet afbrændes i affaldsforbrændingsanlæg.

Det primære fokus i afhandlingen er drivhusgaspåvirkninger (belastninger og besparelser) fra kompostering af organisk affald. Dataindsamling og miljøvurderinger blev udført for centralkompostering af haveaffald (på et fuldskala milekomposteringsanlæg i Aarhus, Danmark) og hjemmekompostering af OHA (i seks hjemmekomposteringsbeholdere som repræsenterede kompostering i en-familie husstande). Afhandlingen præsenterer emissionsfaktorer (EF) i kg gas Mg^{-1} vådt affald for drivhusgasser som udledes i løbet af komposteringsprocessen. Disse emissionsfaktorer er baseret på de mest omhyggelige målekampagner, som til dato er udført for central og hjemmekompostering af organisk affald. Drivhusgasemissionerne blev rapporteret som 2,4 kg CH_4 Mg^{-1} vådt affald og 0,06 kg N_2O Mg^{-1} vådt affald, hvilket giver en total EF på 111 kg CO_2-ækvivalenter Mg^{-1} vådt affald fra centralkompostering af haveaffald. Emissionerne fra centralkompostering blev målt med en 'dynamisk fane' målemetode, som kombinerer en kontrolleret sporgasemission med koncentrationsmålinger i forureningsfanen. Det blev konkluderet, at koncentrationsmålinger i kompostporevolumenet samt målemetoder, som kun dækker en lille del af kompostmaterialets overflade ('small-scale' målemetoder såsom fluxkamre), ikke er velegnede til måling af drivhusgasser fra milekompostering af haveaffald.

Drivhusgasemissioner fra hjemmekompostering af OHA blev også målt (fra en-familie husstande), til dette formål blev der brugt statiske fluxkamre. Disse blev fundet mere brugbare til dette formål, da de dækker hele komposteringsbeholderen. Seks komposteringsbeholdere (med forskellig beluftningsfrekvens) blev fulgt igennem et års kompostering. Emissionerne blev rapporteret som 0,4-4,2 kg CH_4 Mg^{-1} vådt affald og 0,30-0,55 kg N_2O Mg^{-1} vådt affald, med de højeste emissioner fra de beholdere som blev beluftet oftest (hver uge) og de laveste emissioner fra de beholdere, som slet ikke blev beluftet. Disse tal svarer til 100-239 kg CO_2-

ækvivalenter Mg^{-1} vådt affald. Den komposteringsbeholder som repræsenterede den mest realistiske behandling (lav beluftningsfrekvens og lavt input af affald) havde en EF på 127 kg CO_2-ækvivalenter Mg^{-1} vådt affald. Dette er i samme niveau som den EF som blev rapporteret fra centralkompostering af haveaffald, det skal dog bemærkes, at affaldstypen var forskellig i de to tilfælde.

Et andet fokus i afhandlingen, var de miljømæssige besparelser, som er forbundet med brug af kompost i haver. Kompost kan erstatte brugen af kunstgødning og tørv i vækstmedie når det bliver brugt på jord. Denne erstatning er en vigtig parameter ved udførslen af miljøvurdering af kompostering. Den reelle erstatning af disse produkter er dog ikke kendt. Brugerundersøgelser blev udført for at estimere den faktiske erstatning af kunstgødning og tørv med kompost til brug i private haver. Erstatningen blev rapporteret som 41% og 58% for brugerundersøgelsen i Aarhus (n=74) og Københavnsområdet (n=1832), respektivt. Disse resultater indikerede at erstatning af kunstgødning og tørv er langt fra de 100%, som ofte bliver antaget i miljøvurderinger af kompostering.

Emissionsfaktorerne og den estimerede erstatning blev, sammen med andre indsamlede data (for eksempel forbrug af elektricitet og diesel) for centralkompostering af haveaffald og hjemmekompostering af OHA, samlet i livscyklusopgørelser. Disse livscyklusopgørelser blev brugt som udgangspunkt for at udføre livscyklusvurderinger (LCV) af de to nævnte systemer. LCV'erne blev udført ved hjælp af LCV-modelleringsværktøjet EASEWASTE, som medregner alle relevante miljøaspekter af affaldsbehandlingen i en tidsperiode på 100 år. LCV'erne leverede et mere fuldstændigt billede af de miljømæssige sider af affaldsbehandlingen, og de vigtigste og mest følsomme processer, som bidrager til miljøpåvirkninger fra kompostering blev identificeret.

LCV'en af behandling af haveaffald i Aarhus viste at miljøpåvirkningerne fra kompostering var små. De vigtigste processer blev identificeret som værende drivhusgasemissioner (miljøbelastning) og erstatningen af kunstgødning og tørv i vækstmedie (miljøbesparelse). Vurderingen viste at yderligere miljøbesparelser kan opnås ved at føre dele af haveaffaldet (større trædele, som har høj brændværdi og lavt askeindhold) til et affaldsforbrændingsanlæg med energigenvinding. Da det ikke er alt haveaffaldet, som skal brændes, anbefales det at opretholde komposteringsanlægget, så der stadig vil være moden kompost der kan bruges som erstatning for kunstgødning og tørv. Der er et potentiale for at optimere anlæggets

miljøprofil ved at øge brugen af kompost, som erstatning for kunstgødning og tørv samt at optimere komposteringsprocessen for at nedsætte de direkte drivhusgasemissioner. Miljøpåvirkningerne fra hjemmekompostering af OHA var også ret små og de vigtigste processer blev identificeret som værende drivhusgasemissioner samt potentialet for at erstatte kunstgødning og tørv med kompost i havebrug. Vurderingen viste at hjemmekompostering, i flere miljøpåvirkningskategorier, klarede sig lige så godt eller bedre end forbrænding og deponering af OHA (miljømæssigt set). En undtagelse var i kategorien drivhusgaseffekt, hvor forbrænding klarede sig bedst på grund af erstatning af fossil brændsel (kul-baseret energi produktion).

Generelt set er miljøpåvirkninger fra kompostering af organisk affald lave. LCV resultaterne indikerede at hjemmekompostering burde anbefales som en supplerende behandlingsmetode for dele af OHA. Udover besparelserne som blev inkluderet i LCV'erne af kompostering, er der en række fordele som er svære at kvantificere, men som bør tages med i betragtning når man laver miljøvurderinger af kompostering i forhold til alternative behandlingsmetoder. Borgerne er glade for ordningen med at hente kompost og generelt synes de at jorden får en flottere farve ved brug af kompost. Desuden er besparelser af næringsstoffer (især fosfor), bedre jordstruktur og bedre vandholdende evne blot nogle af de supplerende fordele som man får fra kompostering af organisk affald.

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1 Introduction

1.1 Generation of organic waste

Organic waste in the municipal waste stream consists mainly of garden waste and organic household waste (OHW). The term biowaste is sometimes used for the combination of these two types of organic waste. Garden waste consists of woody material, grass clippings, branches, leaves, soil etc. and OHW is the biodegradable fraction of household waste, which is mainly food waste (fresh and cooked). Garden waste makes up a significant proportion of waste in Denmark. In 2006, 598,000 tonnes of garden waste was collected in Denmark, which is equivalent to 18% of the municipal solid waste (MSW). The garden waste generation rate more than doubled from 67 kg person^{-1} year^{-1} in 1994 to 143 kg person^{-1} year^{-1} in 2006. The doubling of the generation rate is mostly a result of better waste collection systems and the improved collection of data, rather than an actual increase in waste generation (Boldrin, 2009). Statistics on garden waste are very limited in a European context, especially due to the fact that many countries only report the total amount of biowaste.

In Denmark, around 1.7 million tonnes of household waste is produced every year (Miljøstyrelsen, 2008). OHW constitutes around 40% of this waste (Petersen and Domela, 2003), although only around 5% of it is treated biologically (ECN, 2010); most is incinerated in waste-to-energy (WtE) plants in Denmark (Miljøstyrelsen, 2008). The total generation of biowaste (including food and kitchen waste from restaurants, caterers, retail premises and processing plants) in the EU is estimated at 118-138 million tonnes year^{-1}, of which 88 million tonnes is municipal waste. It is expected that these amounts will increase by an average of 10% by 2020 (European Commission, 2010). More waste generated, better collection schemes and less waste to landfills will result in larger amounts of organic waste that needs to be treated biologically (e.g. by composting) in the foreseeable future.

1.2 Treatment of organic waste

Organic waste generated in Denmark is handled in different ways. Garden waste is treated almost solely in central composting plants; the share of collected garden waste treated in this way was 99% in 2006 (Miljøstyrelsen, 2008). Recently, it was suggested that some parts of the garden waste should be incinerated in WtE plants in order to utilise energy and thereby reduce the use of

fossil fuels and improve the carbon footprint of waste management systems. This proposal was also motivated by the fact that an updated version of the Biomass Ordinance became effective in January 2010 (in Denmark). In this executive order, garden waste is now categorised as biomass and can thus be incinerated with energy utilisation without tax payment (Miljøministeriet, 2010). Consequently, some incineration plants in Denmark are willing to receive large amounts of garden waste. Environmental comparisons of the treatment of garden waste in either composting plants or WtE plants are also being assessed outside Denmark. Kranert et al. (2010) concluded in such an assessment in Germany that the two treatment methods were of equal value in terms of reducing CO_2 emissions. Other environmental impacts should, however, be considered before making any conclusions on environmental performance.

Only a very small fraction of the OHW generated in Denmark is being composted (25,000 tonnes in 2001; Petersen and Hansen, 2003). In 2001, there were 142 central composting plants in the country, most of which (123) treated garden waste exclusively, while only nine treated OHW together with garden waste (the remaining ten plants also treated sludge and manure). Around 1.1 million tonnes of organic waste (including garden waste, sludge and manure) was composted and 462,000 tonnes of compost produced in 2001. The main use of the compost (45%) was in private gardens (Petersen and Hansen, 2003), but other important uses included agriculture (19%), green areas (10%), landfill covers (6%) and the professional gardening sector (5%) (Petersen and Hansen, 2003).

Small amounts of organic waste never reach the municipal waste stream, but instead are composted at home. According to Petersen and Kielland (2003), the total amount of households that undertake home composting (or backyard composting as it is sometimes called) in Denmark totals 404,000 households (and 61,000 vacation homes). The total yearly estimated amount of OHW that is composted at home is 20,800 tonnes OHW (Petersen and Kielland, 2003). Furthermore, some garden waste is composted at home, but there are no official statistics on these amounts. In a study performed in west London, UK (Smith and Jasim, 2009), the researchers reported that up to 20% of OHW can be diverted away from the municipal waste stream, which means that even with a well-functioning system for home composting, it is still a supplementary technology because the majority of OHW needs to be treated centrally.

The benefits of composting are linked to resource recycling. When composting, carbon (C) and nutrients (N, P and K) may be recycled and used again in soils. Composting can potentially help to restore organic matter in soils, reduce the use of mineral fertilisers and peat in growth media, reduce the need for pesticides, improve soil structure, reduce erosion and improve the water holding capacity of soil. Furthermore, the process can help increase the diversion of organic waste away from landfills. The EU landfill directive was introduced to divert organic waste away from landfills (CEC, 1999), but around 40% of biowaste is still being landfilled in the EU (up to 100% in some EU countries) (European Commission, 2010). A larger proportion of this organic waste will thus be diverted away from landfills in years to come, and more waste will then become available for alternative treatment routes (such as e.g. composting).

1.3 Environmental assessment of composting

Local considerations (e.g. climate, infrastructure and the need for heat, electricity and compost) should be taken into account when choosing either recycling (e.g. composting) or energy utilisation (e.g. anaerobic digestion (AD), incineration, gasification) for the treatment of OHW. For instance, one technology might seem obvious in some parts of the world, whereas it would be unthinkable in other parts. The environmental performance of different waste technologies should be assessed by using solid, site-specific life cycle inventory (LCI) data. LCIs cover all consumptions and emissions of environmental relevance (ISO, 2006a). The number of scientific research studies on the LCI of composting (and especially for home composting) is limited due to difficulties in determining crucial parameters. Generally, the most important processes that contribute to environmental loads (or savings) in relation to composting are gaseous emissions from the degradation of organic waste (loads), as well as savings in emissions related to the use of compost. However, these processes have not previously been assessed comprehensively from full-scale composting plant perspectives, and only partly from home composting. In particular, data has been lacking on the potential savings from the use of compost on soil, which means that previous environmental assessments (including life cycle assessments (LCAs)) have not fully assessed the loads and savings from composting.

Several LCIs have been made for central composting, but often default values (or no values at all) are used for crucial parameters such as greenhouse gas (GHG) emissions, previously estimated or measured from composting at different scales,

but most often for lab-scale or small-scale composting operations (Beck-Friis et al, 2001; He et al., 2001). Measurements have also been performed from pilot-scale composting (full-scale windrow dimensions, but not windrows in operation) (Amlinger et al., 2008; Beck-Friis et al, 2000; Hellebrand, 1998; Hellmann et al., 1997). In these cases, gases were measured with flux chambers or open emission chambers or estimated from concentration measurements in the compost pore space, and no validation of gas emission measurement methods was performed. Beck-Friis et al. (2000) and Hellmann et al. (1997) did not report any total emissions. Table 1 summarises a number of studies where direct GHGs from composting at different scales have been measured or estimated.

Komilis and Ham (2004) and ROU (2006) made comprehensive LCIs of windrow composting, but none of them included CH_4 emissions during composting. ROU (2006) included some downstream savings such as reduced water, fertiliser and herbicide use (processes related to the use of compost), but did not assess the potential of substituting peat with compost.

Home composting has been covered previously in a few scientific studies. Amlinger et al. (2008) undertook a comprehensive study on GHG emissions from home composting, and more complete LCIs of home composting were performed by Colón et al. (2010) and Martínez-Blanco et al. (2010). However, in these studies, important parameters such as leachate emissions and downstream processes were not included. Two of the differences between the home composting studies performed previously and investigations in the present study are differences in the size of the home composting units and the amounts of input waste to the units. Amlinger et al. (2008) described a system with a volume of 0.8 m^3 and weekly additions of waste of up to 53 kg OHW. In the study by Colón et al. (2010) the volume was 0.5 m^3 and weekly input was 18 kg OHW (in comparison to 2.6-3.5 kg $week^{-1}$ in this study). These high-input systems are more likely to represent multiple family home composting instead of single family composting, which is the most common home composting scenario in Denmark.

LCAs should cover all environmental impacts (loads and savings) directly from the waste treatment and indirectly from processes associated with the treatment. Those parameters that are not well covered in the literature (especially the downstream processes that result in environmental savings) should be thoroughly

investigated, and the LCA methodology should be constantly improved to ensure that all possible impacts are included.

Table 1. Review of the most comprehensive studies of GHG emission quantification performed at different composting scales. GW-garden waste; OHW-organic household waste.

Reference	**Scale**	**Amount of material in experiment**	**Input waste material**	**Method applied for the quantification of GHG emissions**
Hellebrand (1998)	Pilot-scale (and transport modelling)	14.8 Mg 115m^3 material*	GW	Modelled from concentration measurements
Beck-Friis et al. (2000)	Pilot-scale	150m^3 material**	OHW	Measured with flux chambers
Hellmann et al. (1997)	Pilot-scale	31.5 Mg 75m^3 material***	OHW (40%) and GW (60%)	Measured with flux chambers
Amlinger et al. (2008)	Pilot-scale	108 m^3 material****	GW	Measured with static open emission chambers
Beck-Friis et al. (2001)	Lab-scale	200 L	OHW and wheat straw	Measured in reactor
He et al. (2001)	Lab-scale	18 L	Food waste and biochips	Measured in reactor
Amlinger et al. (2008)	Home-scale (large input)	1.8-2.9 Mg 800 L	OHW and GW	Open dynamic emission chamber
Martínez-Blanco et al. (2010)	Home-scale (large input)	137 kg 505 L	OHW and GW (0.8:1; volume based)	Calculated from concentrations and air velocity

*estimated from windrow sizes of 3 m high, 7 m wide and 11 m long.

**estimated from windrow sizes of 2.5 m high, 6 m wide and 20 m long.

***estimated from windrow sizes of 1.5 m high, 2.5 m wide and 40 m long.

****estimated from windrow sizes of 2.2 m high, 8.2 m wide and 12 m long.

1.4 Aim of the study

The main aim of this PhD study is to account comprehensively for all impacts (loads and savings) to global warming (GW) and especially measure GHG emissions produced by the composting of organic waste. Two treatment systems

were assessed in detail, namely central windrow composting (full-scale) for the treatment of garden waste, and home composting of OHW.

Since it is not enough only to assess the contributions associated with GW, a secondary aim of the study was to assess fully the two treatment systems and make complete environmental assessments (in a LCA-context) based on the collected data. All contributions to potential impacts were assessed by comprehensive data collection based on real-site field investigations. The data was gathered in LCIs and further used as input data for the environmental assessments (LCA) of composting and alternative treatment options for organic waste.

Detailed, site-specific data is very important as input into LCAs, as it decreases uncertainty and increases the specificity and consistency of the assessment. The thesis includes specific and accurate waste composition, and shows how Material Flow Analysis (MFA) and Substance Flow Analysis (SFA) can be used to increase understanding of the system under study. Apart from the estimation of emission factors (EFs) (in kg gas Mg^{-1} wet waste (ww)) for the direct emissions of GHG, the study provides estimates of the actual substitution of fertiliser and peat in growth media when compost is used in private gardens, as well as the significant potential for improving environmental performance by optimising the use of compost.

The quantification of GHG emissions from central composting and home composting is presented in Andersen et al. (I) and Andersen et al. (III), respectively. Full LCIs of these two types of composting are assessed in Andersen et al. (II) and Andersen et al. (IV). Boldrin et al. (V) present an account of all contributions to global warming from composting and compost use, which is also assessed in Andersen et al. (VI) in a user survey study on the substitution of peat and fertiliser when using compost in private gardens. Finally, the environmental assessments (LCAs) of garden waste composting (full-scale) and OHW composting (home-scale) are presented in Boldrin et al. (VII) and Andersen et al. (VIII).

2 Quantification of greenhouse gas emissions from composting

GHG emissions and GW have recently become very important topics on the political agenda, especially after the Kyoto protocol (in 1997) and more recently the release of the fourth assessment report (AR4) (in 2007) from the Intergovernmental Panel on Climate Change (IPCC). There is now international consensus that something has to be done in order to minimise the increase in global temperature (i.e. GW) caused by the anthropogenic release of GHGs. Contributions to GW from composting (and all other waste management options) are small compared to the total contributions from all sectors (including the energy sector), but there is a large potential to mitigate GHGs from the waste sector. In relation to composting, the GHGs of importance are methane (CH_4) and (N_2O), the global warming potentials (GWPs) of which are 25 and 298, respectively, meaning that they are 25 and 298 times more potent than fossil carbon dioxide (CO_2) (Solomon et al., 2007). Thus, CH_4 and N_2O have the potential to contribute significantly to GW, even in the event of relatively low emissions. Biogenic CO_2 emitted during composting is most often regarded as neutral with regards to GW, since the CO_2 emitted has recently been taken up by the biomass (the GWP of biogenic CO_2 is zero as stated by Christensen et al., 2009).

The composting process and especially temperature development during the composting process are described in Section 2.1, following which the actual quantification of GHG from windrow composting of garden waste (Section 2.2) and home composting of organic household waste (Section 2.3) is outlined.

2.1 Composting process

Composting is an aerobic process in which organic matter is degraded by the microorganisms present in organic material. Outputs of the composting process are (mainly) CO_2, water, minerals and biologically stabilised material, which is collectively referred to as ‘compost’ (Stentiford and Bertoldi, 2010). Composting usually takes place in central facilities in the form of either open (e.g. windrow, static pile, mattress), enclosed (e.g. channel and cell, aerated pile) or reactor (e.g. tunnel, rotating drum) technologies. The most common waste fractions composted are garden waste, OHW, sludge and manure - and most often a mixture of these fractions. In addition to these central facilities, home

composting (of OHW and garden waste) is a preferred treatment option in some countries, especially in the EU.

One of the main differences between central and home composting is the lower additions of waste and lower temperatures for home composting, as the temperature development increases significantly during central composting and is often above 70°C. As easily degradable organic matter is degraded, microbial activity decreases and so does temperature (Stentiford and Bertoldi, 2010). In contrast, the temperature in home composting is normally not elevated due to the low volume of material (Alexander, 2007). Examples of temperature development during windrow composting (Andersen et al., I) and home composting (Andersen et al., III) are presented in Figure 1. The measurements were performed in the compost pore space, approximately 1 m below the waste surface in central composting and 30 cm below the waste surface in home composting. The temperature increased significantly during the first weeks in the central composting plant (Aarhus, Denmark) and reached approximately 80°C after 200 days of composting. Turning events (indicated by dotted lines and arrows) facilitated a lower temperature, especially towards the end of composting (Andersen et al., I). In the case of home composting, the temperature did not exceed 30°C at any time during composting and was always 1-10°C higher than the ambient temperature (Andersen et al., III), even after the addition of waste stopped (indicated by the dotted line on the right side of Figure 1).

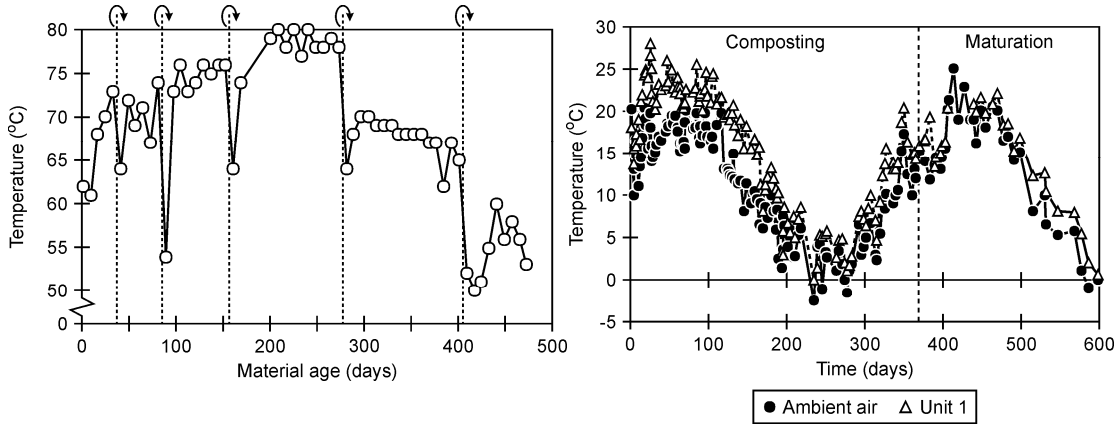

Figure 1. Left: Temperature development during windrow composting of garden waste. The dotted lines and arrows represent turning events (Andersen et al., I). Right: Temperature development during home composting of OHW. The graph shows the development in Unit 1 during the composting and maturation phase. The two phases are distinguished from each other by a dotted line (Andersen at al., III).

2.2 Greenhouse gas emissions from windrow composting of garden waste

Windrow composting (open composting technology) is the most common technology used for the treatment of garden waste in Denmark. In total, 131 of the 142 composting plants in existence in the country in 2001 were of the windrow type (Petersen and Hansen, 2003). The composting facility in Aarhus is a typical Danish windrow composting plant with limited control and management and processed 16,220 tonnes of garden waste in 2007 (reference year for the assessment). Garden waste was delivered at the facility, shredded and then organised in elongated piles (windrows) of 115 m in length, 9 m in width and 4.5 m in height. The newest material was placed in the western part and the oldest material was situated in the eastern part of the facility. The material was turned (in an eastern direction) approximately every second month with a front loader. There was no addition of water and no collection of leachate and gases. The entire process lasted for 12-14 months and the mature compost was sieved and prepared for the market.

2.2.1 Compost gas composition

The production of CO_2, CH_4, CO (carbon monoxide) and N_2O is a result of the microbial degradation of organic material (CO is not a direct GHG, but was included since the instrument was setup to measure it). Before measuring the emission of these gases from the windrow composting plant in Aarhus, Denmark, the concentrations of gases inside the windrows were mapped. This was done by analysing compost air that was withdrawn from probes driven into the compost material (in the compost pore space). Spatial variability was assessed by taking samples from nine points in cross-sections from different places in the windrow system, representing different ages of material. Figure 2 shows the average values (for 14 cross-sections) in L L^{-1} for O_2, CO_2 and CH_4 and µL L^{-1} for CO and N_2O for each of the nine points in the cross-sections, as described in Andersen et al. (I).

The concentrations of oxygen (O_2) and CO_2 show clearly that microbial activity is taking place. O_2 concentrations are highest on the edges of the windrows (up to 12 vol% around 1m inside) and lower towards the centre (4-8 vol%) whereas CO_2 concentrations are highest in the centre. In addition, CH_4 is produced in significant concentrations, especially in the centre of the material. Up to 22 vol% on average was found from the 14 cross-sections (the highest concentration

measured was 44 vol% CH_4). This is in good accordance with findings by Beck-Friis et al. (2000), who established that CH_4 concentrations increase along with an increase in windrow height. In addition, the turning of the material at the Aarhus composting plant was only performed every second month, allowing only limited diffusion of O_2 into the material, which resulted in very long periods of optimal conditions for methanogenic bacteria. N_2O concentrations were also highest in the centre, though slightly better distributed than CH_4, which could be because N_2O is a by-product of both nitrification and denitrification (Eggleston et al., 2006) and is produced mainly in anaerobic pockets where an O_2 gradient is present (Beck-Friis et al., 2000). CO concentrations were quite evenly distributed in the cross-sections with a tendency to higher concentrations towards the edges, which is in accordance with findings by Hellebrand and Kalk (2001) and Hellebrand and Schade (2008), who reported that CO occurs when O_2 is present.

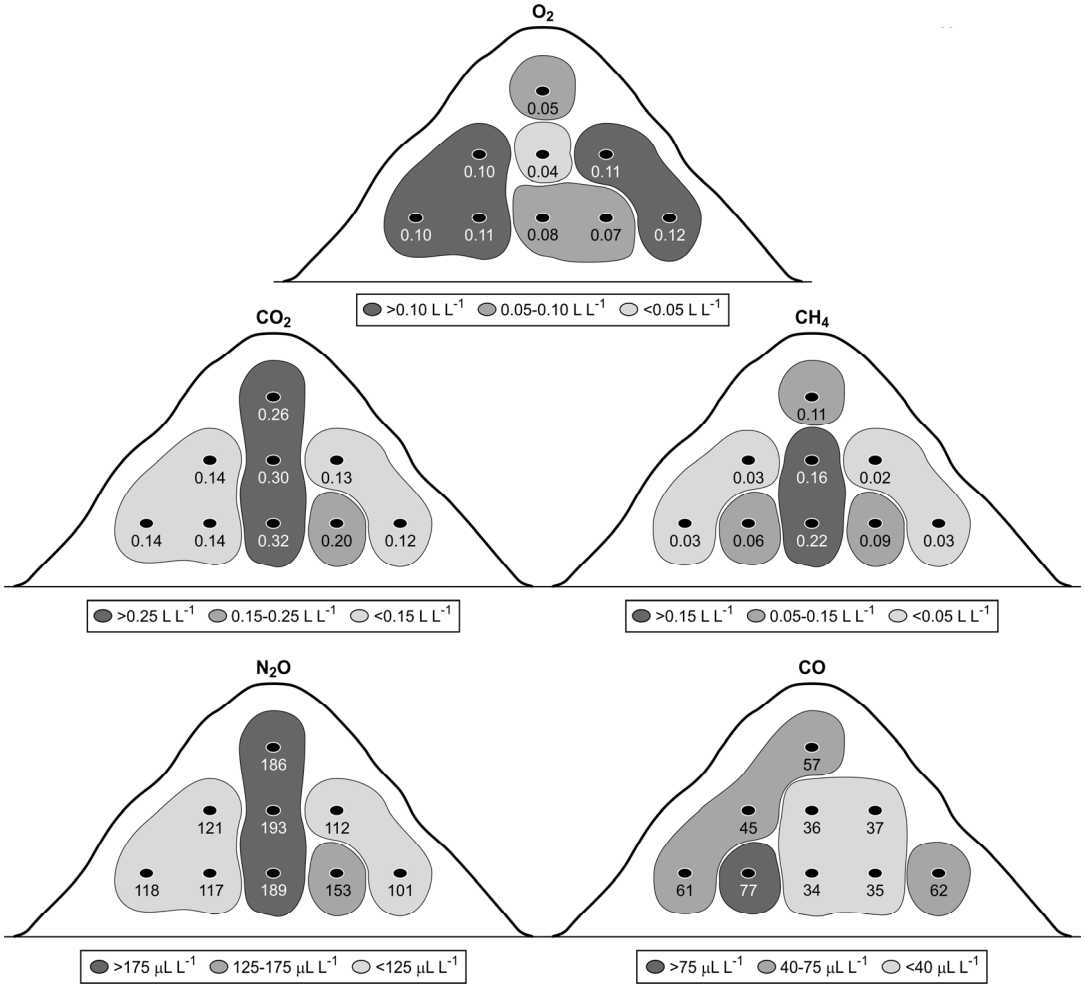

Figure 2. Cross-sectional distribution of O_2, CO_2 and CH_4 (in L L^{-1}) and CO and N_2O (in µL L^{-1}) in the compost pore space. The numbers are average concentrations of 14 cross-sections in the windrow system (Andersen et al., I).

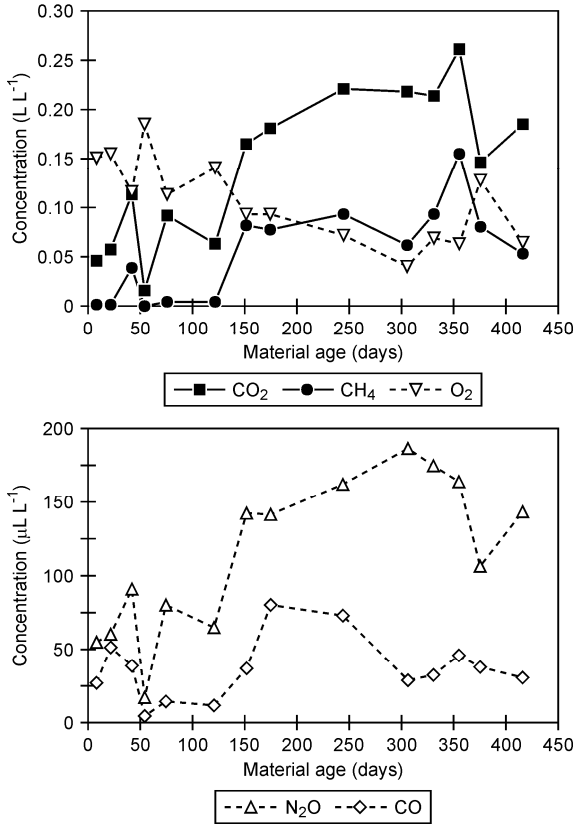

Figure 3. Mean concentration of O_2, CO_2 and CH_4 (top graph, in L L^{-1}) and CO and N_2O (bottom graph, in µL L^{-1}) in the compost pore space during windrow composting of garden waste. Each point is a mean value of the nine points in each cross-section (Andersen et al., I).

Gas composition in the windrows varied over time as presented in Figure 3. Each point in the figure is an average of the nine points in each cross-section. In general, O_2 was highest in the early stages of composting (12-19 vol%) and decreased during the process to around 4-7 vol%. Concentrations of GHGs increased in the same period, while CO_2, CH_4 and N_2O concentrations peaked between 200 and 350 days of composting. The peak concentration for CO was observed somewhat earlier (after 180-250 days). The concentration of GHGs generally decreased towards the end of composting (after 400 days).

2.2.2 Quantification of GHG emissions from windrow composting

The governing flow of air in windrow composting was examined using flux chambers placed as shown in Figure 4 (where T denotes top, W denotes west and E denotes east). It was found that around 50% of the measured gas fluxes (CO_2, CH_4, CO and N_2O) were captured from the flux chamber at position T1, while 85-100% of the fluxes were captured at positions T1-T3 (flux results are shown

in the bottom part of Figure 4). These results indicated that most gases escape in a narrow (1 m wide) area in the top of the composting windrow. The effect of air coming in through the lower parts of the windrow and leaving in the top (as outlined in the top of Figure 4) is referred to as the 'chimney-effect'. This phenomenon has previously been described by Hellmann et al. (1997) and Poulsen (2010) among other authors.

Concentrations of GHGs in the windrow system were mapped, and the results indicated that the organic waste produced significant quantities of GHGs, suggesting that windrows may serve as a source for GHG emissions into the atmosphere. However, the emission of gases depends on a range of processes (including CH_4 oxidation) and the actual emissions need to be measured directly at the facility. GHG emissions were measured at the full-scale windrow composting plant in Aarhus (treating garden waste) using two small-scale methods and a total emission measurement method. In addition, a transport model described by Hellebrand (1998) was tested. The measurements were performed during 2007 and 2008.

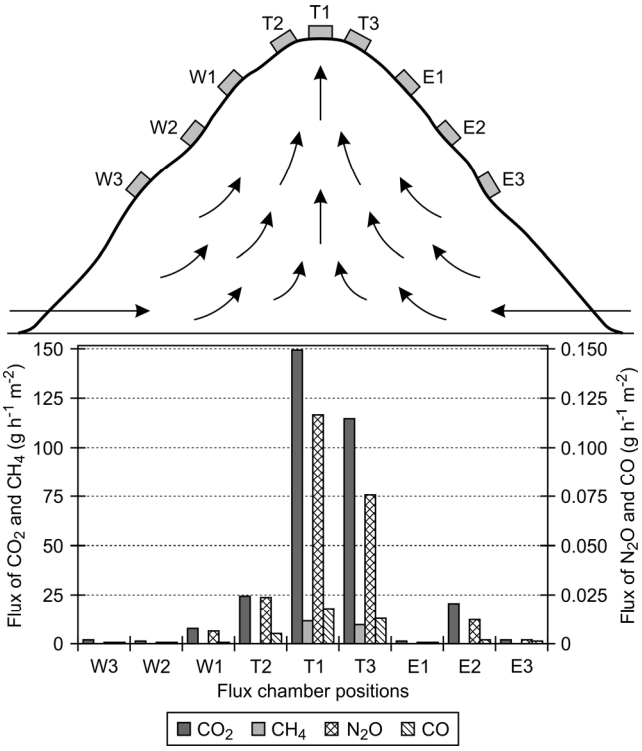

Figure 4 Conceptual drawing of the 'chimney effect'. Gases are drawn in at the sides of the windrow and emitted through the top centre. The monitoring points of the small flux chambers are shown on top of the windrow, and the flux results are presented in the bar diagram under the drawing. 'T' denotes top, 'E' denotes east and 'W' denotes west.

The first small-scale method applied on-site was a static flux chamber method, described by Beck-Friis et al. (2000), Börjesson and Svensson (1997), Livingston and Hutchinson (1995) and Scheutz et al. (2003, 2007) among others. In this method, the flux chambers are placed on top of the windrows (from where the gases are emitted) and gases increase in the chambers due to diffusion (and other gas transport) processes. The flux was measured over a total timeframe of six minutes, in which a linear increase of gases was recorded. The second small-scale method was a funnel method, in which the air flow and (constant) concentration of gases were measured (an average of approximately 20 minutes of continuous measurements were recorded). In both methods a flux of gases (in g h^{-1} m^{-2}) from the top of the windrows was translated into an emission factor (in g h^{-1}) by extrapolating the results from the measurements to an estimate from the whole composting plant. Pictures of both small-scale methods are presented in Figure 5. The numbers presented in Table 2 (in Mg yr^{-1}) have been rescaled to represent an annual release of gases.

Figure 5. Pictures of the two small-scale measurement methods used at the composting plant in Aarhus, Denmark, namely the static flux chamber (left) and the funnel method (right).

Instead of measuring the gases locally at the windrow, a total emission method measures the emissions from a distance by integrating the emission plume from

the entire facility. The total-emission method applied at Aarhus composting plant is a dynamic plume method (Börjesson et al., 2009; Mellqvist, 1999), which combines controlled tracer gas release along with concentration measurements in the downwind plume. In this way, the entire emission from the facility is measured during relatively short campaigns. Two pictures of the tracer gas release and the measurement equipment used in the dynamic plume method, respectively, are presented in Figure 6. A transport (gas) model (Hellebrand, 1998) was used to check the use of concentration measurements (in the compost pore space) for an estimation of emissions from composting facilities. All methods have been described in detail and the results were compared to each other and to the transport model in Andersen et al. (I).

Table 2. Mean concentration in compost pore space (in ppmv) and calculated/measured emissions (in Mg yr^{-1}) including standard deviations using the three different measurement methods and the transport model (Andersen et al., I).

	CO_2-C	**CH_4-C**	**N_2O-N**	**CO-C**
Mean concentration in compost pore space (ppmv)*	176900	67400	138	51
Flux chamber (Mg yr^{-1})	107 ± 30	16 ± 6.1	0.44 ± 0.03	0.08 ± 0.06
Funnel (Mg yr^{-1})	71 ± 23	7.6 ± 0.54	0.25 ± 0.09	0.07 ± 0.03
Dynamic plume (Mg yr^{-1})	1748 ± 210	38 ± 7.6	1.0 ± 0.46	2.4 ± 1.1
Transport model (Mg yr^{-1})	999	381	1.8	0.29

*The transport model does not include a method to calculate standard deviations. The uncertainty is related to concentration measurements.

Table 3. Emission estimates (in % of degraded element mass) from field campaigns performed between 2007 and 2008. The numbers are average numbers including standard deviations from two (flux chamber and dynamic plume) and three field campaigns (funnel) respectively.

	Emission (% of degraded element mass)			
	CO_2-C	**CH_4-C**	**N_2O-N**	**CO-C**
Flux chamber	7.7 ± 2.2	1.2 ± 0.4	10 ± 1.6	0.006 ± 0.005
Funnel	5.1 ± 1.7	0.6 ± 0.04	5.7 ± 2.0	0.005 ± 0.004
Dynamic plume	127 ± 15	2.7 ± 0.6	23 ± 11	0.17 ± 0.08
Transport model*	72	28	42	0.02

*The transport model does not include a method to calculate standard deviations. The uncertainty is related to concentration measurements.

Figure 6. Pictures of the dynamic plume method. Left: Tracer gas release on top of the windrow. Right: Fourier Transform Infra-Red (FTIR) measurement system incorporated in a van.

Emission estimates from the three methods and the gas transport model are presented in Mg yr^{-1} in Table 2 and as a percentage of the lost element mass (C and N) in Table 3. The variation in emission estimates between the methods was significant, the main conclusion of which is that the small-scale methods significantly underestimated the gaseous losses, whereas the dynamic plume method slightly overestimated the losses. Only 6 to 9% of the C degraded during composting could be accounted for using the small-scale methods, whereas the dynamic plume method estimated that 127% and 3% of the lost C was emitted as CO_2 and CH_4, respectively (see Table 3). Variations in fluxes, using the small-scale methods were significant, even within very short periods of time (the results of flux dynamics are presented in Andersen et al. (I)), and thus these methods are not suitable for measuring GHG emissions from windrow composting of garden waste through small-scale methods. The dynamic plume method was assumed the most reliable, and the estimate of 127% of lost carbon emitted as CO_2 thought to be quite a good estimate. The carbon balance was thought to have low uncertainty (many sampling campaigns during the year, see Section 3.1.1), whereas the uncertainty for gas emission measurements was

higher due to problems representing seasonal variations (due to only two measuring campaigns).

From the measurements derived from the dynamic plume method, it was calculated that the EF for windrow composting was 2.4 kg CH_4 Mg^{-1} ww and 0.06 kg N_2O Mg^{-1} ww, which is equivalent to 111±30 kg CO_2-equivalents (eq.) Mg^{-1} ww (using GWPs of 25 and 298 for CH_4 and N_2O, respectively; as stated by Solomon et al., 2007). These estimates are higher than what Amlinger et al. (2008) suggest for a well managed composting plant, which is 20-65 kg CO_2-eq. Mg^{-1} ww. Emissions reported where in the same level as reported by other authors. Hellebrand (1998) estimated (with a gas transport model) EFs of 6.7 kg CH_4 Mg^{-1} ww and 0.08 kg N_2O Mg^{-1} ww, which is equivalent to 194 kg CO_2-equivalents (eq.) Mg^{-1} ww for windrow composting of garden waste. Amlinger et al. (1998) reported EFs (with open emission chambers) of 68 kg CO_2-equivalents (eq.) Mg^{-1} ww for windrow composting of garden waste.

Calculations carried out according to a simple transport model (Hellebrand, 1998) based on compost pore space concentrations, predicted an unexpectedly high emission of CH_4 (10 times greater than observed with the dynamic plume method). The calculated emissions did not include, for example, diffusion and CH_4 oxidation, as inclusion of the former would have increased predicted CH_4 emission, whereas inclusion of the latter would have reduced this factor. These limitations made the emission estimate less robust, and the transport model was thus considered insufficient for the prediction of CH_4 emissions.

One of the reasons why the small-scale methods were not found suitable for measuring gaseous emissions was that the variations, even within a short distance and short time, were significant. A difference factor of 9-10 in fluxes (for all gases) in a period of three days at the same position at the composting plant was observed with the funnel method (Andersen et al., I). In addition, it was believed that the natural flow of air through the windrow system (the chimney effect) could possibly be disturbed when chambers are placed on top of the compost material. The gases might thus escape around the chamber instead of vertically into the chamber, resulting in significant underestimations. Livingston and Hutchinson (1995) state that underestimations of gas fluxes are very likely when employing flux chambers on highly porous material (such as garden waste), due to leakages of the enclosed air. Thus it was found that the dynamic plume method

(total-emission method) was the most suitable method for measuring the GHG emissions from windrow composting of garden waste.

2.3 Greenhouse gas emissions from home composting of organic household waste

The most commonly used home composting units in Denmark were chosen for the experimental home composting setup. Six units, each of 320 L, were set up and monitored during one year of composting, plus an additional three months of maturation (where no waste was added). A schematic drawing and picture of the composting units are presented in Figure 7. The six composting units were fed with OHW from different families, meaning that input waste was different in all composting units. The home composters were supplied with a guideline that encouraged not to compost meat, dairy products and any non-biological waste. Waste added during the entire year varied between 115 and 184 kg OHW, which is equivalent to 2.6-3.5 kg per week (see input for each unit in Table 4).

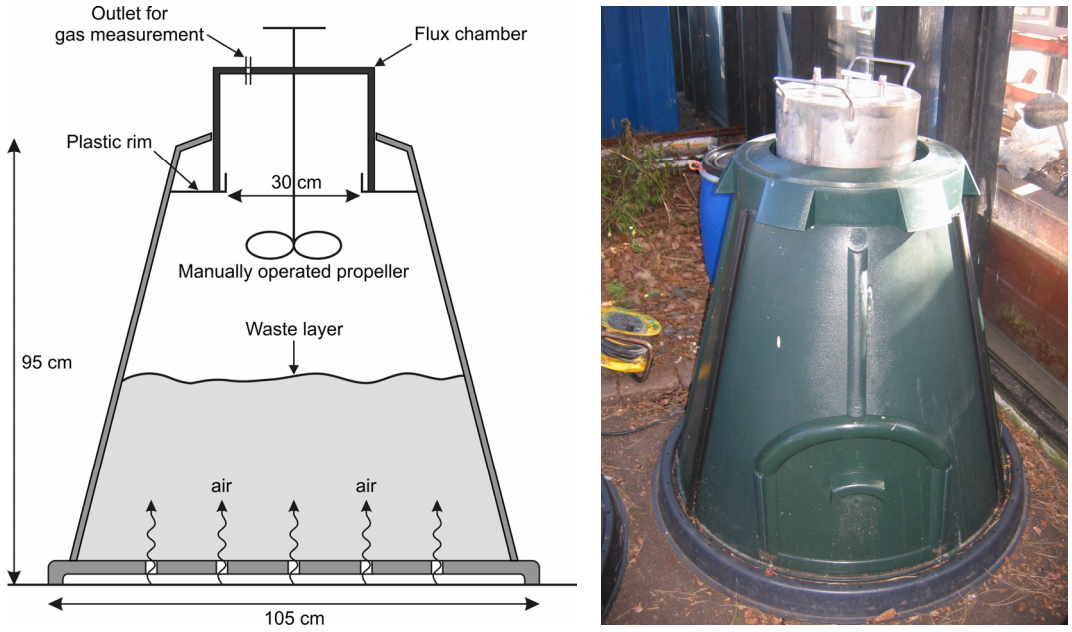

Figure 7. Left: Schematic drawing of one of the composting units including the position of the flux chamber system during measurement campaigns. The possible flow of air through the bottom of the composting unit is shown with arrows on the drawing. Right: Picture of one of the composting units.

The most important parameter believed to affect GHG production and emissions the most was the mixing frequency, which thus varied between the units. Units 1 and 2 were mixed every week (frequent mixing), Units 3 and 4 were mixed every

sixth week (infrequent mixing) and Units 5 and 6 were not mixed at all (no mixing).

The approach to measuring GHG emissions from home composting was quite different than for the windrow composting system, and involved a static flux chamber system attached on top of the composting units. As opposed to the measurements from windrow composting, the home composting units were covered entirely by the flux chamber, and thus it was easier to catch the entire flux of gases. The emission measurements (in kg h^{-1}) were performed over a short period of approximately six minutes per measurement (with linear increase of gases in the chamber) and were carried out twice a week for a year of composting. Additionally, emission measurements were performed across a three-month maturation period, although with lower frequency.

Emission measurements (in kg h^{-1}) are presented as a function of time in Figure 8 for all six home composting units, where the dotted lines distinguish between the composting and the maturation period. Emission variation over time and between composting units was significant, as there was a tendency for increasing GHG emissions during the spring/summer periods and decreasing emissions during winter (around 200-300 days of composting). This was most evident for Unit 1 - emissions decreased slowly after the end of the composting period (dotted line), but did not stop completely until after the three months of maturation. Any extra emissions observed after the addition of waste was stopped are referred to as 'tail-emissions'.

The increased emission in Unit 4 (and to a lesser extent in Unit 5) was due to high-load experiments across a three-month period after the year of composting. Unit 4 was fed with high loads of OHW (up to 25 kg OHW every fortnight) and Unit 5 with high loads of garden waste (around 3 kg every fortnight), to examine how increased waste input would affect temperature and GHG emissions. The temperature was not highly affected by the additional inputs of waste (temperatures were never above 30°C). This is contrary to what was reported by e.g. Amlinger et al. (2008) who measured temperatures of up to 70°C from high-input home composting.

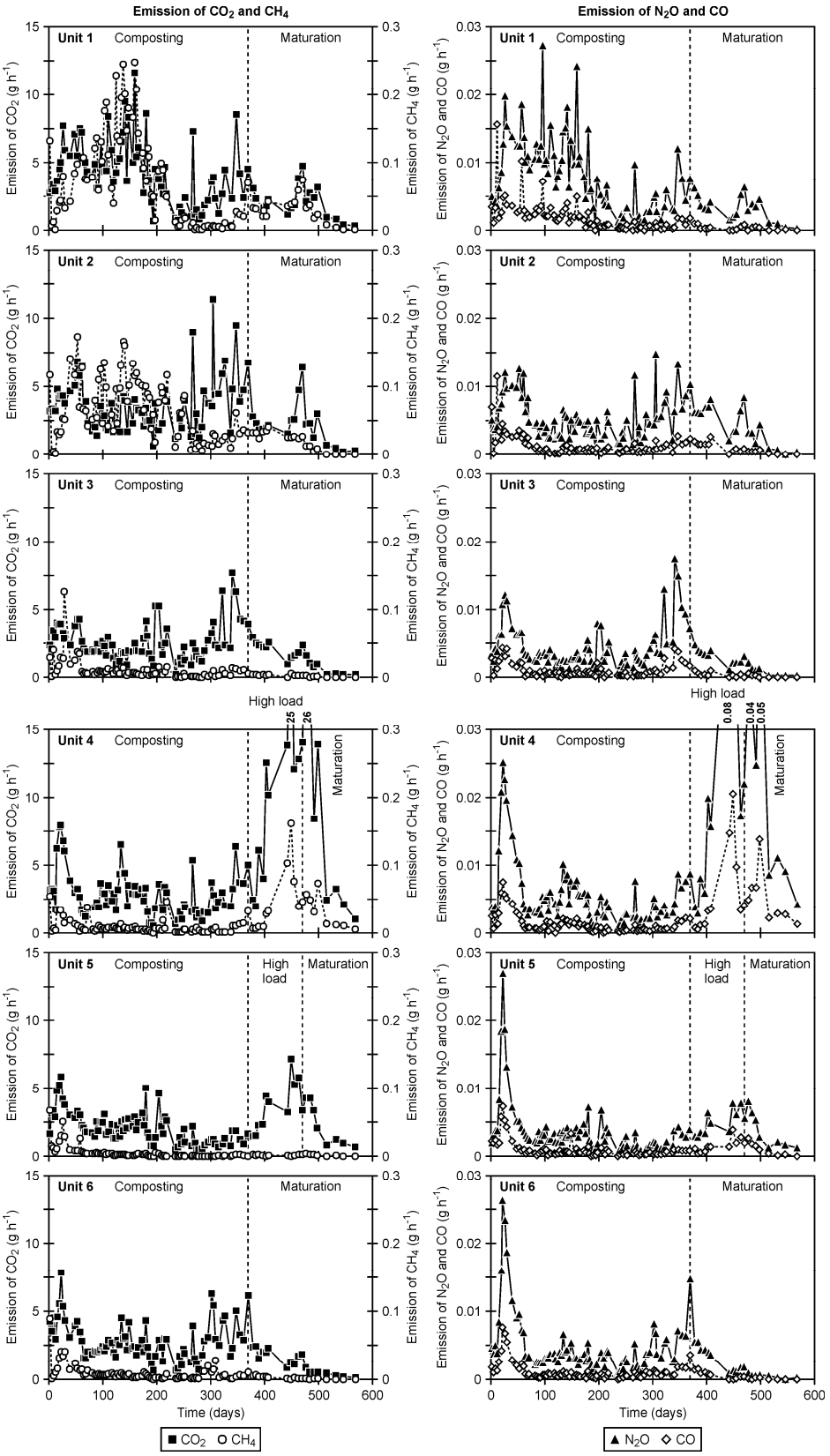

Figure 8. Emissions (in g h^{-1}) of CO_2 and CH_4 (left column) and N_2O and CO (right column) in Units 1-6 during home composting and maturation. The different phases are distinguished from each other by dotted lines (Andersen et al., III).

Total emissions of each gas were estimated by integrating the emission curves in Figure 8, which resulted in the calculation of the total mass loss of each gas. The total emitted mass (in kg Mg^{-1} ww) for each of the composting units is presented in Table 4. This was estimated at 177-236 kg CO_2 Mg^{-1} ww, 0.4-4.2 kg CH_4 Mg^{-1} ww, 0.30-0.55 kg N_2O Mg^{-1} ww and 0.08-0.13 kg CO Mg^{-1} ww, which is equivalent to 100-239 kg CO_2-eq. Mg^{-1} ww (accounting only CH_4 and N_2O). Besides these EFs, an additional emission of CH_4 was observed from the frequently mixed units (Units 1 and 2) during mixing of the material. This was estimated as an extra 8-12% on top of the reported EFs (Andersen et al., III). The emissions increased significantly during the high-load experiment, indicating that higher input of waste entails higher emissions of GHGs. This was primarily the case in Unit 4, where additional OHW was added, and only to a minor extent in Unit 5, where additional garden waste was added (see Table 4).

The reported EFs were in the same range as given by Amlinger et al. (2008) and Martínez-Blanco et al. (2010). In a home composting study (large composting units of 0.8 m^3) with high-input OHW (up to 53 kg per week) performed by Amlinger et al. (2008), the EFs were measured as 76-187 kg CO_2-eq. Mg^{-1} ww. This was based on emissions of 0.8-2.2 kg CH_4 Mg^{-1} ww and 0.19-0.45 kg N_2O Mg^{-1} ww. Martínez-Blanco et al. (2010) reported emissions of 0.16 kg CH_4 Mg^{-1} ww and 0.68 kg N_2O Mg^{-1} ww, equivalent to 207 kg CO_2-eq. Mg^{-1} ww. In the study by Martínez-Blanco et al. (2010), the amount of input waste was also significantly higher (18 kg per week), possibly resulting in a different rate of mineralisation.

From the comprehensive quantification campaigns from both central and home composting, it can be concluded that GHG emissions are in the same order of magnitude. However, the emissions from home composting (mainly CH_4) are higher than from central plants if the composting units are mixed very frequently (every week). It should be noted that EFs from central and home composting cannot be compared directly, since the input material is different. Furthermore, the presented EFs for central and home composting only include the direct emission related to the microbial degradation of organic matter. All other contributions (direct and indirect) to environmental loads should be included when undertaking environmental assessment of waste treatment systems, which is covered in Section 3.3 about LCIs.

Table 4. Weekly input to Units 1-6 and emission factors (in kg Mg^{-1} ww) from home composting of organic household waste in Unit 1-6. Total global warming emission factors are calculated in the last column (in kg CO_2-equivalents Mg^{-1} ww) (Andersen et al., III).

		Emission factors (kg Mg^{-1} ww*)				
Unit number	**Input (kg $week^{-1}$)**	**CO_2**	**CH_4**	**N_2O**	**CO**	**Total EF (CO_2-eq.)**
1	3.5	252	4.2	0.45	0.10	239
2	3.4	240	3.7	0.39	0.09	210
3	2.8	209	0.8	0.36	0.08	127
4	2.9	236	1.0	0.55	0.13	187
5	2.2	177	0.4	0.30	0.08	100
6	3.2	189	0.6	0.32	0.07	111
4 (only composting)**	-	175	0.7	0.34	0.08	117
4 (high load + tail-emission)**	-	306	1.3	0.78	0.19	265
5 (only composting)**	-	156	0.4	0.29	0.08	95
5 (high load + tail-emission)**	-	295	0.1	0.41	0.11	125

*ww, wet waste

**The lower part of the table shows the emission factors for each phase (composting and high load phase including tail-emissions) in Unit 4 and 5.

3 Life-cycle inventories and the modelling of composting technologies

The LCI part of an LCA study involves the modelling, compilation and quantification of inputs and outputs regarding a defined system. This is the most time-consuming task in an LCA since it involves a great deal of work to quantify the flows and emissions of the system under study (Wenzel et al., 1997).

This section of the thesis is divided into four parts. First, the material flow analysis (MFA) and substance flow analysis (SFA) are described (Section 3.1), followed by outcomes from user surveys on compost use (Section 3.2). This data, together with GHG emission data (from Section 2), is compiled in the LCIs of central and home composting (Section 3.3). Potential ways of assessing the environmental performance of composting systems (based on the LCIs) are addressed in Section 3.4 (GHG accounting and LCA modelling).

3.1 Material flow analysis

MFA and SFA were carried out using STAN (short for subSTance flow ANalysis) software, which performs these functions according to the Austrian standard ÖNORM S 2096. MFA and SFA are valuable tools used to evaluate waste systems and to follow flows of material and important substances through a system. STAN can be used to reconcile uncertain data and to compute unknown flows including uncertainties (Cencic and Rechberger, 2008). The MFA and SFA for the two composting systems function in helping to understand the systems, and have provided the basis for LCA modelling in EASEWASTE, focusing on the material (waste), C, nutrients (N, P and K) and heavy metals (As, Cd, Cr, Cu, Hg, Ni, Pb and Zn).

3.1.1 Central composting

The input material (garden waste) at Aarhus composting plant was sampled twice per season (eight times) over a one-year period (2006-2007). The sampling involved an initial sorting of garden waste into five material fractions (small stuff, branches, wood, foreign items and hard materials). The amounts of garden waste sampled weighted between 14 to 31 tonnes in all sampling campaigns. These large samples underwent a four-step mass reduction scheme (Boldrin, 2009) according to representative sampling techniques (Gy, 1998), in order to derive 5 g laboratory samples. The sampling campaign was tested and validated, after

which it was concluded that the method was robust and adequate enough to characterise the waste (Boldrin, 2009). The compost was much more homogenous than the input material due to degradation and mixing, and was sampled with a large number of grab increments, five times during a year. Approximately 70 kg of compost was sampled and mass reduced according to a two-step mass reduction scheme (Boldrin, 2009), to produce 5 g laboratory samples. The other flows, e.g. recirculated material (see Figure 9), were sampled once only. The laboratory samples were sent for chemical analysis at an external certified laboratory (ALS Scandinavia AB, Luleå, Sweden).

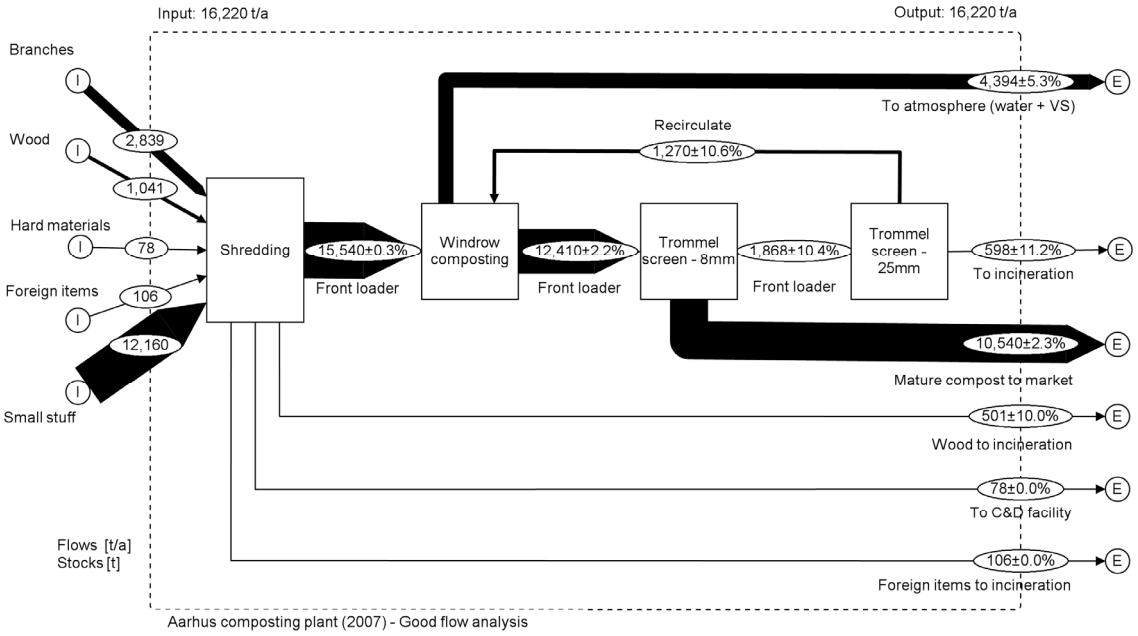

Figure 9. Material flow analysis (in Mg wet waste) of windrow composting of garden waste at Aarhus composting plant. All numbers are in Mg material yr^{-1} based on wet weight (Andersen et al., II).

The MFA (and system boundaries) of the Aarhus composting plant is presented in Figure 9. Most of the input material (68%) ended up in the mature compost, whereas 28% was lost to the atmosphere as moisture and volatile solids (VS). The SFAs are not shown here, but can be seen in Andersen et al. (II). The main results of the SFAs showed that the C (and VS) loss to the atmosphere was 56% while 37% was left in the compost. The calculated N loss to atmosphere was less than 1% but uncertainty was large. Heavy metals were mainly found in the compost, but all concentrations were still below accepted threshold values for application on soil. Based on this and other parameters (C/N was 16.1), it was concluded that the compost from Aarhus composting plant was of high quality

and thus useful in horticulture or in private gardens as a substitute for fertilisers and/or peat in growth media.

3.1.2 Home composting

In the home composting setup input material was sampled with a large number of grab increments (1% of the input based on mass) before each addition of waste (twice a week), from each of the six composting units (Andersen et al. IV). The compost was sampled, mass reduced and prepared for analysis according to representative sampling techniques (Gy, 1998). The 5 g laboratory samples were sent for chemical analysis at an external certified laboratory (ALS Scandinavia AB, Luleå, Sweden).

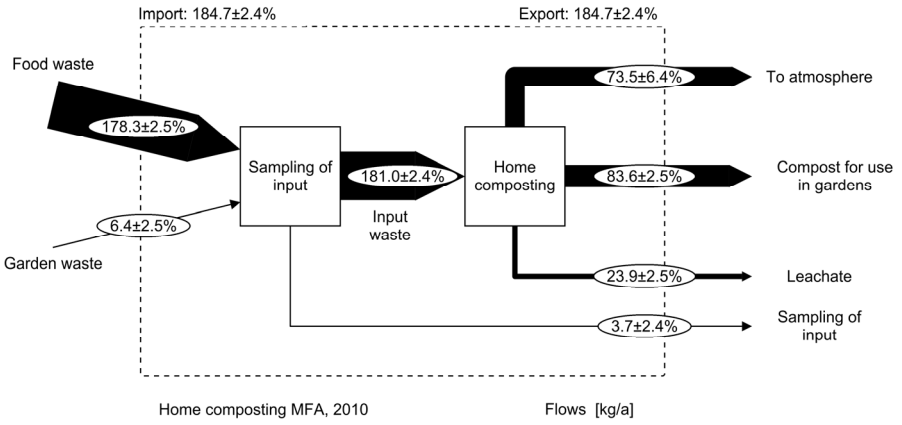

Figure 10. Material flow analysis of home composting of OHW in Unit 1. All numbers are in kg material yr^{-1} based on wet weight (Andersen et al., IV).

The MFA (and the system boundaries) of the home composting system, represented by Unit 1, is shown in Figure 10. In this case, 185 kg of OHW (including garden waste) was composted during one year. The loss of material (including water) in the six composting units was 55-73% (55% in Unit 1) to the atmosphere and via leachate. The results of the SFAs are not shown her, but they are described in details in Andersen et al. (IV). The loss of VS, C and N was 66-79% (71% in Unit 1), 63-77% (68% in Unit 1) and 51-68% (51% in Unit 1), respectively. The SFAs furthermore showed that nutrients and heavy metals were found mainly in the compost and thus there were no significant losses via air and leachate. The concentrations of heavy metals in the compost were below all threshold values for application on soil (Andersen et al. IV). The C/N ratio of the produced composts from Units 1-6 was 15.8-18.0 and the compost composition was generally within the range previously reported in the literature. Thus, the

compost was regarded as being suitable as a substitute for inorganic fertilisers and/or peat (in growth media) for use in hobby gardening.

3.2 User survey on compost use

Four user surveys were carried out in order to obtain information on compost use and especially to examine the substitution of peat, fertiliser and manure in hobby gardening. This was done by performing three comprehensive user surveys at a recycling centre (RC) in Aarhus municipality (Surveys 1, 2 and 3), which resulted in a total of 138 answers (n=138), and one user survey in the Copenhagen area (survey 4), which produced 1832 answers (n=1832). In Denmark, garden waste compost is delivered (from the windrow composting plants) to the RCs, from where it is provided (sold or given away) to local citizens.

The purpose of Surveys 1-3 was to obtain general information on compost use (how much is used, what kind of products are used, garden size, distance from the home, etc.) as well as to investigate the substitution of peat, fertiliser and manure (the latter, however, was only assessed in Surveys 2 and 3, n=74). The purpose of Survey 4 was to obtain more answers on the substitution question and thereby obtain a more solid dataset. The general results on the user surveys are not presented here; instead, the focus is on substitution mechanisms. In addition to the user surveys, six home visits were performed as case studies, where the actual application of compost was examined and valuable information on substitution mechanisms observed (Andersen et al. VI). The combination of user surveys and home visits was thought to be the best way to obtain information on substitution of alternative soil improvers. Home visits to more people would be preferable, but due to restrictions of time and resources, this was not possible.

3.2.1 Substitution of peat, fertiliser and manure

The most important part of the user surveys was estimating the substitution of peat, fertiliser and manure when compost was used in private gardens, which would ultimately indicate to what extent the use of peat, fertiliser and manure is actually avoided. The compost users were asked whether they fully (100%), partly (25%) or do not (0%) avoid the use of peat, fertiliser and manure when they use compost in their gardens. The distribution of answers in Surveys 2 and 3 is shown in Figure 11 (the distribution of answers from Survey 4 is presented in Andersen et al. (VI)). The substitution was estimated as 22% for peat, 12% for

fertiliser and 7% for manure in Surveys 2 and 3, while it was 19% for peat, 24% for fertiliser and 15% for manure in Surveys 4. The substitution elements were added to give a total estimate of the substitution. The total substitution was thus estimated as 41% for Surveys 2 and 3 and 58% for Survey 4.

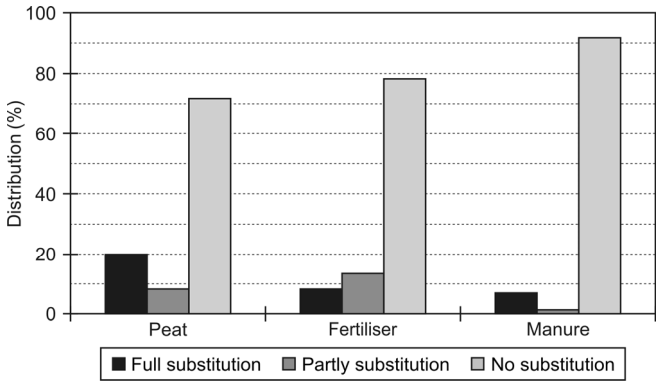

Figure 11. Substitution of peat, fertiliser and manure with compost in Surveys 2 and 3 in Aarhus (n=74). 'Full substitution' means 100%, 'partly substitution' means 25% and 'no substitution' means 0% substitution (Andersen et al. VI).

These results indicate that it cannot be assumed that all compost produced can substitute for alternative products (as sometimes assumed in environmental assessments) and that there is significant potential for increasing the amount of compost that can be used in a more rational way (and thereby receive credits for the environmental savings made). A cautious estimate based on the amounts of compost picked up at the RC in Aarhus, and the substitution estimates was made. It showed that 1000 kg of compost could substitute for 58 kg of peat, 0.2 kg N, 0.2 kg P and 2.1 kg K for fertiliser and 0.1 kg N, 0.1 kg P and 1.3 kg K for manure. Details on the methodology and results are presented in Andersen et al. (VI). The significance of peat and fertiliser substitution in a GW context is covered in Boldrin et al. (V) (see Section 3.4.1).

The case studies (home visits to private gardens in northern Aarhus) showed that some people used their compost for purposes other than as a substitute for peat, fertiliser and manure, instead using it as general soil improver (e.g. to obtain better structure and growth conditions) and as filling material (instead of soil). Some other benefits from using compost in private gardens were also identified from the case studies - it made the garden look nicer (colour) and people were in general happy with the service (picking up compost) that the municipality

provided. Some of these softer issues are difficult to quantify and thus, often not included in LCAs.

3.3 Life-cycle inventories

All of the contributions to environmental loads and savings from the two types of composting systems are summarised in LCIs described in the following sections. Section 3.3.1 covers the LCI of central composting of garden waste whereas Section 3.3.2 covers home composting of OHW. Ammonia (NH_3) is not a GHG and was thus not included in Sections 2.2 and 2.3. It is, however, potentially produced during composting and consequently can affect acidification and nutrient enrichment and it is therefore assessed in Sections 3.3.1 and 3.3.2.

3.3.1 LCI of central composting

All the collected LCI data for windrow composting in Aarhus is presented in Table 5. At the composting plant in Aarhus some energy and material consumptions were reported. Diesel was used for shredding, excavating and moving waste around (1.50 L Mg^{-1} ww) and for turning and screening (1.54 L Mg^{-1} ww). Electricity was used for illumination and the heating of engines (heavy machinery) and administration buildings (0.2 kWh Mg^{-1} ww). In addition, small quantities of lubricating grease (0.013 L Mg^{-1} ww), motor oil (0.005 L Mg^{-1} ww), hydraulic oil (0.005 L Mg^{-1} ww) and cleaning fluids (0.001 L Mg^{-1} ww) were used at the plant. No water was added and no leachate recorded. Smaller amounts of leachate could be generated, but the losses of nutrients and heavy metals were considered to be of minor importance (Andersen et al., II).

The quantification of GHGs is described in detail in Section 2.2.2. NH_3 could not, despite several attempts using both passive sampling with diffusion tubes and active sampling with impinger flasks, be detected with reasonable accuracy (due to very low concentrations of NH_3). The very low concentrations of NH_3 (in the ppb level) indicated that emissions were limited, while the uncertainty in the calculated N balance suggested that at least some of the N could be lost as NH_3, which was identified as one of the most sensitive parameters in the modelling of environmental performance (see Section 4.1.2).

Table 5. LCI data for Aarhus garden waste composting plant (Andersen et al. II).

		LCI data	Amount	Unit
Inputs	Waste	Amount of garden waste	16,220	Mg ww yr^{-1}
	Energy consumption	Electricity	0.2	kWh Mg^{-1} ww
		Diesel	3.04	L Mg^{-1} ww
	Materials consumption	Lubricating grease	0.013	L Mg^{-1} ww
		Motor oil	0.005	L Mg^{-1} ww
		Hydraulic oil	0.005	L Mg^{-1} ww
		Cleaning fluid	0.001	L Mg^{-1} ww
Outputs	Gaseous emissions (to atmosphere)	CO_2-C biogenic	86 ± 10	kg Mg^{-1} ww
			97.6	(% of total C emitted)
		CH_4-C	1.9 ± 0.4	kg Mg^{-1} ww
			2.1	(% of total C emitted)
		N_2O-N	0.05 ± 0.01	kg Mg^{-1} ww
			23	(% of total N emitted)
		CO-C	0.12 ± 0.06	kg Mg^{-1} ww
			0.3	(% of total C emitted)
	Liquid emissions (to groundwater)	Leachate	0	L Mg^{-1} ww
	Products (and rejects)	Compost	649	kg Mg^{-1} ww
		Wood to incineration (screen rejects)	37	kg Mg^{-1} ww
		Wood to incineration (sorting rejects)	31	kg Mg^{-1} ww
		Hard materials to C&D facility	4.8	kg Mg^{-1} ww
		Foreign items to incineration	6.5	kg Mg^{-1} ww

3.3.2 LCI of home composting

Home composting is managed by private garden owners, and in many cases there is very limited control of the composting units. No water was added to the units during composting, since the input waste was already relatively wet. No electricity or any fuels or materials were used during composting. Therefore, in general, the only contributions to environmental loads from the home composting

process were from gaseous emissions and via leachate production (Andersen et al., IV).

Quantification of GHGs is described in detail in Section 2.3. The concentration of NH_3 was measured in all six composting units during a two-month period (mid-November 2008 to mid-January 2009), by using a passive sampling approach. The emission of NH_3 was estimated by assuming that the (linear) relationship between the concentration in the composting unit and the emission of NH_3 was the same as for CO_2. This, of course, added some uncertainty, but it indicated the magnitude of the emissions. The linear relationship between interior concentrations and emissions was quite good for CO_2 (R^2=0.7214) and when assuming the same correlation for NH_3 the estimated loss was 0.03-2.0 g Mg^{-1} ww or what is equivalent to maximum 0.004% of the lost N during composting. These low losses were supported by very low NH_3 concentrations (2-121 ppbv) in the composting units. NH_3 emissions have been reported previously by Amlinger et al. (2008). In this study, an EF of 0.47-0.97 kg NH_3 Mg^{-1} ww was reported from (high-load) home composting. As described by Amlinger et al. (2008), NH_3 is normally produced when the temperature is high (above 40-50°C), which backs up the measured low concentrations of NH_3 in this study (very low temperatures compared to, for example, Amlinger et al. (2008)).

Leachate was collected and sampled during two two-month campaigns in November-December 2008 and March-April 2009 in Unit 1. Leachate quantity (in L Mg^{-1} ww) and composition were assumed to be the same for the other units. The samples were frozen immediately after sampling and later sent for analysis at a certified external laboratory (ALS Scandinavia AB, Luleå, Sweden). The total volume of leachate collected was around 3.7 L for both campaigns, which is equivalent to 130 L of leachate per Mg of input waste. This is in the same order of magnitude as found by other authors. Wheeler and Parfitt (2002) and Amlinger et al. (2008) found a leachate generation of 31 and 270 L Mg^{-1} ww, respectively. Even though the loss of total mass was relatively large through leachate, the losses of nutrients and heavy metals via leachate were found to be insignificant. Losses of C and N via leachate were 0.3-0.6% of the total C loss and 1.3-3.0% of the total N loss during composting in the six composting units. The loss of heavy metals via leachate was considered insignificant and was not considered to constitute a problem. All contributions to environmental loads are presented in Table 6.

Table 6. LCI data for home composting of organic household waste (Andersen et al., IV).

		LCI data	Amount	Unit
Inputs	Waste	Organic household waste	113-273	kg ww yr^{-1}
		Garden waste	6-22	kg ww yr^{-1}
	Energy and materials consumption	Electricity	0	kWh Mg^{-1} ww
		Water	0	L Mg^{-1} ww
Outputs	Gaseous emissions (to atmosphere)	CO_2-C (biogenic)	48-69	kg Mg^{-1} ww
			51-95	(% of total C emitted)
		CH_4-C	0.3-3.1	kg Mg^{-1} ww
			0.3-3.9	(% of total C emitted)
		CO-C	0.03-0.06	kg Mg^{-1} ww
			0.04-0.08	(% of total C emitted)
		N_2O-N	0.08-0.15	kg Mg^{-1} ww
			2.8-6.3	(% of total N emitted)
		NH_3	~0	kg Mg^{-1} ww
	Liquid emissions (to groundwater)	Leachate	130	L Mg^{-1} ww
		N losses	0.05	kg Mg^{-1} ww
			0.3-0.6	(% of total N emitted)
		C losses	0.33	kg Mg^{-1} ww
			1.3-3.0	(% of total C emitted)
		BOD	3.5	kg Mg^{-1} ww
		COD	9.9	kg Mg^{-1} ww
		K	6.4	kg Mg^{-1} ww
		P	0.08	kg Mg^{-1} ww
		As	$2.4 \cdot 10^{-5}$	kg Mg^{-1} ww
		Cd	$2.5 \cdot 10^{-6}$	kg Mg^{-1} ww
		Cr	$3.2 \cdot 10^{-5}$	kg Mg^{-1} ww
		Cu	$2.9 \cdot 10^{-4}$	kg Mg^{-1} ww
		Hg	$2.8 \cdot 10^{-7}$	kg Mg^{-1} ww
		Ni	$8.7 \cdot 10^{-5}$	kg Mg^{-1} ww
		Pb	$9.9 \cdot 10^{-5}$	kg Mg^{-1} ww
	Product	Compost	272-453	kg Mg^{-1} ww

3.4 Environmental accounting and assessment of composting

LCIs can act as a starting point for performance of environmental assessments in different forms. GHG accounting and LCA have been chosen as two possible

ways of assessing environmental performance of waste systems. GHG accounting is dealing with GW contributions only, and can be used for reporting and accounting on a company level or on regional/national level, often in relation to CO_2 taxation. LCA, on the other hand, provides a more holistic view of the environmental performance and can be used as a decision-support tool for waste management companies or for decision-making at municipal/regional/national level. Other methods and tools exist, but have not been the focus in this thesis.

3.4.1 GHG accounting and GW contributions

A full GHG account was made, in order to show all contributions (loads and savings) to GW (Boldrin et al., V). It was made as a literature review (including data from Andersen et al., I) and is divided into open and closed technologies as well as home composting. All contributions have been divided into 'upstream', 'operation' and 'downstream' (U-O-D), as suggested by Gentil et al. (2009). 'Operation' refers to direct emissions from the composting plant, whereas 'upstream' and 'downstream' are indirect processes taking place before and after the actual composting.

The net GHG account presented in Boldrin et al. (V), for open and enclosed technologies, as well as home composting is presented Table 7. The numbers reported for the different technologies (open, enclosed and home composting) include studies of both garden waste and OHW composting. All processes contributing to global warming can be seen in the more elaborate U-O-D tables in Boldrin et al. (V).

In general, upstream emissions (provision of electricity and diesel) contribute little to the overall GHG account, except in some case for enclosed technologies. Direct (Operation) contributions from the composting were moderate depending on the technology. Values of up to 242 kg CO_2-eq. Mg^{-1} ww and 220 kg CO_2-eq. Mg^{-1} ww was reported for open technologies and home composting, respectively, whereas contributions from enclosed technologies were lower (up to 81 kg CO_2-eq. Mg^{-1} ww) due to the treatment of the gases in biofilters. The main potential environmental savings were from the use of the compost (indirect downstream process), especially when assuming that it substitutes peat in growth media (a value of up to -880 kg CO_2-eq. Mg^{-1} ww was reported). The environmental savings reported were lower for substitution of fertilisers (a value of up to -145 kg CO_2-eq. Mg^{-1} ww). It is noteworthy that the benefits from using compost are

calculated as potentials and that these are far from being actualised, which was shown in the user surveys on compost use (Section 3.2). The ranges of reported values are significant, especially for the downstream processes, which emphasises that precise, local data (real measurements) should be used when undertaking environmental assessments. There are a number of different ways to reduce the direct GHG emissions, but the real challenge is to utilise the compost in a way that gives the most benefits.

The net GHG account for two cases of windrow composting of garden waste and home composting of OHW (as reported in Andersen et al. (IV) and Boldrin et al. (V)) are presented in the two bottom rows of Table 7. The savings in CO_2 from peat substitution were in the low end of ranges presented by Boldrin et al. (V), namely -180 and -109 kg CO_2-eq. Mg^{-1} ww for central composting of garden waste and home composting of OHW, respectively.

Table 7. Net greenhouse gas account for different composting technologies divided into 'upstream', 'operation' and 'downstream' (U-O-D) contributions. All numbers are expressed as global warming factors (GWFs) (in kg CO_2-eq. Mg^{-1} ww). The two bottom rows contain examples of the two modelled cases of windrow composting of garden waste and home composting of OHW.

	Upstream (indirect)	Operation (direct)	Downstream (indirect)
	GWF (kg CO_2-eq. Mg^{-1} ww)		
Open technologies* Boldrin et al. (V)	0.2 to 20	3 to 242	-145 to +45 (fertiliser) -880 to +44 (peat)
Enclosed technologies** Boldrin et al. (V)	1 to 60	5 to 81	-145 to +45 (fertiliser) -880 to +44 (peat)
Home composting Boldrin et al. (V)	0	77 to 220	-146 to +17 (fertiliser) -880 to +44 (peat)
Open composting*** Boldrin et al. (V)	1.2	119	-14 to -32 (fertiliser) -180 (peat)
Home composting**** Andersen et al. (IV)	0	100 to 239	-4 to -20 (fertiliser) -67 to -109 (peat)

*Open technologies include windrow, static pile and mattress composting

**Enclosed technologies include aerated pile, tunnel and drum composting

***In this case, open composting refers to windrow composting (of garden waste)

****Home composting of OHW

3.4.2 LCA modelling of waste systems

LCA is an internationally standardised method for carrying out environmental assessments (ISO, 2006a; ISO, 2006b). LCAs were originally made for determining the environmental performance of products. However the LCA methodology can be used for the environmental assessment of waste management (which is last stage of a traditional LCA).

LCA has become a common tool for the assessment of solid waste management systems in recent years (Kirkeby et al., 2006b). The advantage of using this approach on solid waste management systems is that it covers all impacts associated with waste management as well as processes upstream and downstream from the actual waste system. Upstream and downstream processes such as energy and material production are often very important, and can in many cases exceed the direct impacts of the waste management system itself (Ekvall et al., 2007). LCAs are often used as a decision support tool for planning waste management systems and for policy-making. There is, however, a range of uncertainties and limitations associated with LCA modelling, such as choice of time perspective, choice of system boundaries and general assumptions.

The LCA method applied for the environmental assessments described in Section 4 was the EDIP1997 method (Wenzel et al., 1997), and the waste-LCA-tool EASEWASTE was used. EASEWASTE was developed for estimating waste flows, resource consumption and environmental emissions from waste management systems. A complete LCA within a 100-year time horizon can be carried out, including a range of potential impact categories such as global warming (GW), photochemical ozone formation (POF), acidification (AC), nutrient enrichment (NE), ecotoxicity in soil (ETs), ecotoxicity in water (ETw), human toxicity in soil (HTs), human toxicity in water (HTw) and human toxicity in air (HTa). The first four impact categories are collectively referred to as 'non-toxic' (or energy-related), while the remaining impact categories are referred to as 'toxic'.

EASEWASTE operates with 48 waste material fractions, each described by 40 components, and includes several modules for modelling specific technologies for handling, treating or disposing of waste. The technologies modelled in EASEWASTE were windrow composting and home composting ('Biotechnology module'), incineration ('Thermal treatment' module) and

landfilling (‘Landfill mixed waste’ module). The general concept of the EASEWASTE model is described in Kirkeby et al. (2006b) and the modules and sub-modules in Riber et al. (2008), Manfredi and Christensen (2009), Hansen et al. (2006) and Boldrin et al. (2010b). The actual modelling of organic waste management is presented in Section 4 as two cases of windrow composting of garden waste and home composting of OHW, respectively. The results for the environmental impact categories were normalised and presented in the unit ‘person equivalents’ (PE), whereby 1 PE expresses the average environmental impact from one average person in one year in the given impact category.

4 Environmental assessment of composting technologies

Full environmental assessments of composting were performed for windrow composting of garden waste and for home composting of organic household waste in an LCA-context in the following sections.

4.1 Environmental assessment of garden waste management in Aarhus, Denmark

The environmental assessment of garden waste management was part of a larger project for the Aarhus municipality, the objective of which was to identify key environmental loads and savings and to produce recommendations on improving the environmental profile of garden waste management.

4.1.1 Modelling and assumptions

The functional unit of the LCA was defined as the management of garden waste treated at Aarhus composting plant during 2007 (16,220 Mg), and the time horizon for the environmental assessment was 100 years. The system boundaries were expanded to include all (indirect) upstream and downstream processes associated with garden waste treatment, according to the U-O-D concept (Boldrin et al. V; Gentil et al., 2009). The system boundaries of current management practices are shown in Figure 12.

The assessment is based on waste composition and LCI for central composting, as described in Sections 3.1.1 and 3.3.1. NH_3 emissions were not quantified, but were assumed to be 6.6% of the total N loss (Boldrin et al., VII; based on Amlinger and Peyr, 2002). Six scenarios were assessed based on the treatment of 16,220 Mg of garden waste (ww). The current management of garden waste (scenario 1) was assessed together with five alternative scenarios including windrow composting (all scenarios), incineration in WtE plants (scenarios 2-6) and home composting in private gardens (scenarios 5-6). Detailed descriptions of the scenarios and modelling in EASEWASTE are presented in Boldrin et al. (VII), while the system boundaries for all scenarios can be seen in Boldrin et al., 2009.

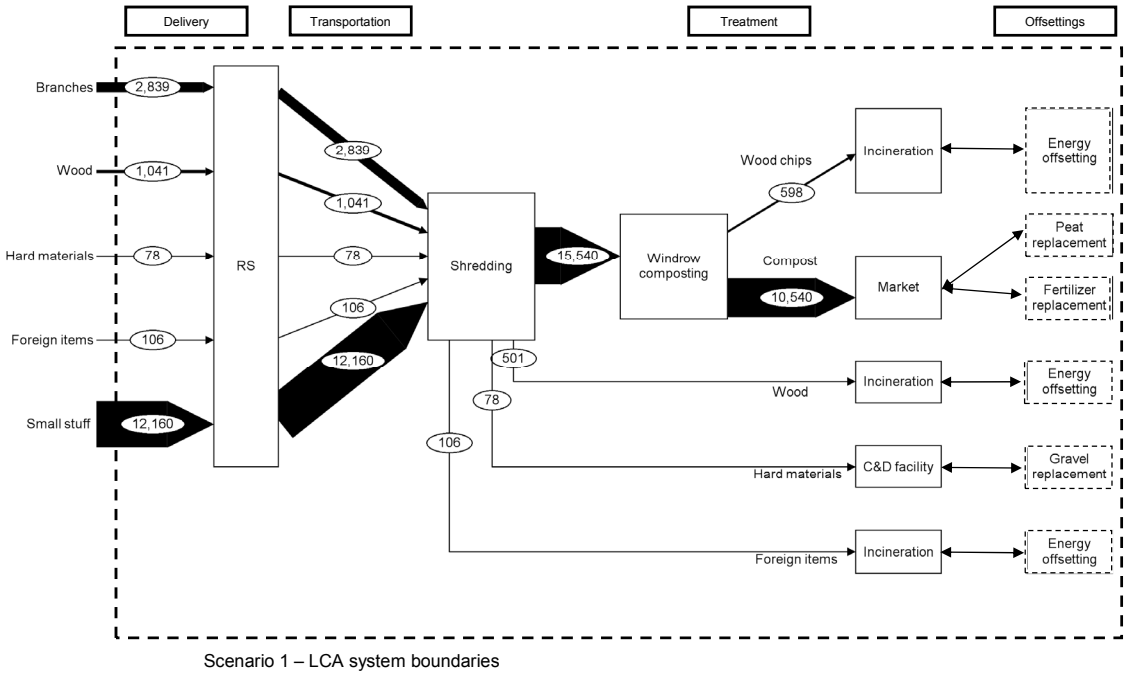

Figure 12: LCA system boundaries for scenario 1 (the current garden waste management in Aarhus). Material flows are expressed in Mg of ww (Boldrin et al., VII).

- ***Scenario 1****. Current management:* Central windrow composting. Foreign items, large woody parts and screen residues (woodchips) (>25 mm) were incinerated (Figure 12).
- ***Scenario 2.*** *Composting and incineration of rejects:* All screen residues (>8 mm) were incinerated. The rest was composted as in scenario 1.
- ***Scenario 3.*** *Composting and seasonal incineration:* Garden waste received during winter (December, January and February) was incinerated. The rest was composted as in scenario 1.
- ***Scenario 4.*** *Maximum incineration:* Garden waste received during winter (December, January and February) and all screen residues (>8 mm) were incinerated. The rest was composted as in scenario 1.
- ***Scenario 5.*** *Home composting:* 25% of the fraction 'small stuff' was home composted in private gardens. The rest was composted as in scenario 1.
- ***Scenario 6.*** *Home composting and maximum incineration:* 25% of the fraction 'small stuff' was home composted in private gardens. Garden waste received during winter (December, January and February) and all screen residues (>8 mm) were incinerated. The rest was composted as in scenario 1.

The LCA was performed by using EASEWASTE and the following assumptions were made: The compost produced in Aarhus is substituted for peat and used in the preparation of commercial growth media and as a replacement for inorganic fertilisers. Peat substitution was modelled on a 1:1 volume basis (Boldrin et al., 2010a), and it was assumed that 50% of the compost used in hobby gardening actually substituted for peat, which was based on preliminary user survey results (Boldrin et al., 2009) (elaborate user survey results were later reported in Andersen et al. (VI)). Fertiliser substitution was modelled based on the nutrient content of the compost and the nutrient utilisation rate (the fraction of nutrients that can replace inorganic fertilisers, which depends on the availability of the nutrients). Here, also a 50% substitution was assumed. When the compost is used on soil, some carbon is still bound to the soil at the end of the 100-year time frame, which is credited to the system as avoided CO_2 emissions and the value estimated at 14% of the total carbon content in the compost (Bruun et al., 2006). Home composting was modelled by assuming 40% degradation of VS and with gaseous emissions according to the work by Amlinger et al. (2008) (the results by Andersen et al. III and Andersen et al. IV were not ready at the time of the assessment). The gaseous emissions were, however, measured from OHW in the work by Amlinger (2008) and not from garden waste as in the modelled assessment, which adds some uncertainty to the numbers.

The energy (electricity and heat) generated at the WtE incinerator in Aarhus (located at the same waste facility as the composting plant) was assumed to substitute for marginal energy production technologies, namely coal-based electricity and coal-based heat. The WtE facility is equipped with a grate furnace with a combined heat and power (CHP) energy recovery system. Cleaning of the flue gas was done with semidry and wet systems, and activated carbon was used for the removal of dioxin and Hg. NO_x was removed by Selective Non-Catalytic Reduction (SNCR) system. The efficiency of the plant was 20.7% for electricity production and 74% for heat production (calculated on LHV). LCI for the WtE plant used in the modelling is presented in Riber et al. (2008).

4.1.2 Results

The main source of environmental impacts (positive PE values) from the current management scenario (scenario 1) derives from the actual composting process. Contributions to GW come from GHGs (CH_4 and N_2O) from degradation of the organic material and from the combustion of fuel (fossil CO_2) in heavy

machinery at the facility. NH_3 volatilisation and NO_x emissions from the combustion of fuel are the main contributors to NE and AC, whereas NO_x, volatile organic compounds (VOCs) and CO contribute to POF. The main contributions to the toxic potential impact categories are from polycyclic aromatic hydrocarbons (PAH), due to combustion of fossil fuels especially in ETw, and from the use of compost in gardens, due to chromium and arsenic (in HTs) and mercury (in HTw) contained in the compost. The largest savings (negative PE values) to the system are from the substitution of peat by compost, especially in terms of GW. The credit is due mainly to the avoidance of fossil CO_2 emissions during peat degradation. Savings were also seen from fertiliser substitution in the potential impact category NE (savings in phosphorous emissions), HTs (savings in chromium emissions) and HTw (savings in mercury emissions) due to avoided production of P fertiliser.

Figure 13 shows potential impacts from the six garden waste management scenarios. All impacts were added in to single impact categories, and so the figures thus show the net load or saving. In general, impacts arising from the treatment of garden waste were relatively small compared to the treatment of other types of MSW (Kirkeby et al., 2006a), as the environmental loads were -6 to 8 mPE Mg^{-1} ww and 4 to 100 mPE Mg^{-1} ww for the non-toxic and toxic impact categories, respectively. Toxic impacts are, however, most likely overestimated since all heavy metal concentrations were below threshold values for compost use on soil, and thus no recommendations should be based on the results of the potential toxic impact categories alone (the LCA methodology accounts for toxic effects based on the amounts - not concentrations - of heavy metals). The issue regarding potential toxic impact categories is discussed further in Section 5.2.

Most of the alternative scenarios (scenarios 2-6) performed environmentally better than the current scenario (see Figure 13). The introduction of incineration and/or home composting of parts of the garden waste seemed to potentially improve the system. Home composting (scenarios 5 and 6) had only small potential benefits, mainly due to the avoidance of collection and transportation. The incineration of parts of the garden waste, on the other hand, resulted in large improvements in most impact categories. The most significant impact was in the GW category, produced by avoiding the production of electricity and heat from fossil fuels (coal) when incinerating the garden waste (especially scenarios 3, 4

and 6). The high ash content (48.3% on average during the year; Andersen et al. II) and the low LHV (6.4 GJ Mg^{-1} ww on average during the year; Andersen et al. II), suggest that only parts of the most woody material (the fractions 'branches' and 'wood') are suitable for incineration. Therefore, it is not recommended to incinerate all garden waste.

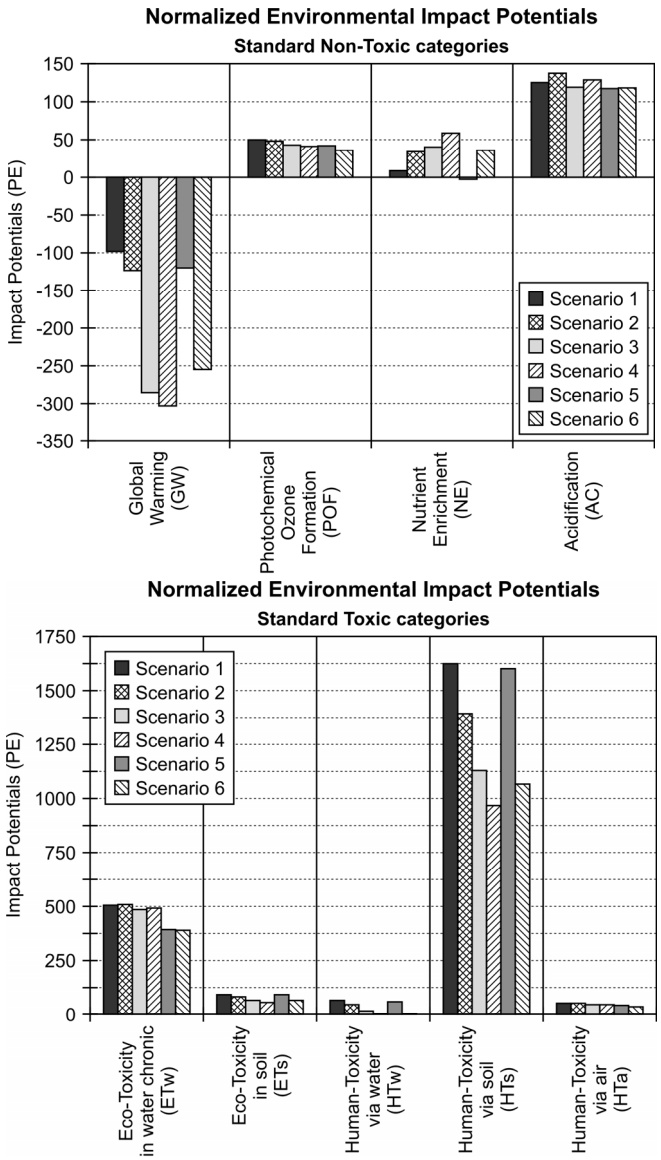

Figure 13. Comparison of potential non-toxic (top) and toxic (bottom) environmental impacts for the current management of the Aarhus composting plant (scenario 1), as well as alternative scenarios (scenarios 2-6) (16,220 Mg of garden waste) (Boldrin et al., VII).

An uncertainty analysis of the assessed system (where the systems were modelled with changes in key parameters) showed that the most critical

parameters were the chosen value for peat substitution (which was selected from a preliminary user survey on compost use at the time of modelling environmental performance) and the N degradation. Peat substitution was, as described in Section 3.2.1, probably overestimated in the LCA on garden waste treatment. In addition, N losses would be better covered by measuring total NH_3 emissions from the facility.

The assessment of garden waste treatment in Aarhus showed that it is environmentally favourable to incinerate parts of garden waste that exhibit high LHV and low ash content (the woody parts), which could act as start-up material for the ovens or as extra input material when heat is needed (in winter). The composting process should, nevertheless, still be maintained in order to produce mature compost for use in gardens, in order to increase the substitution of peat and thus increase environmental performance significantly. The assessment showed that home composting is environmentally similar to the alternative treatment options. In some potential impact categories there are benefits from composting parts of garden waste at home.

4.2 Environmental assessment of home composting

The objective of the LCA of home composting was to assess the environmental profile of home composting of OHW, and identify the key environmental loads and savings during the process. It was also the aim to compare performance of alternative treatments such as incineration and landfilling.

4.2.1 Modelling and assumptions

The functional unit of the LCA was defined as the management of 1 Mg of OHW, and the time horizon and the system boundaries were set as in the LCA on garden waste management (100 years and all upstream and downstream processes included). The assessment was based on the waste compositions and LCI data presented in Sections 3.1.2 and 3.3.2 (Andersen et al., IV). All scenarios assessed the treatment of 1 Mg of OHW. Detailed descriptions of the scenarios and the modelling in EASEWASTE are presented in Andersen et al. (VIII).

- ***Scenarios 1 and 2.*** *Home composting, frequent mixing:* Home composting with mixing of the material every week.
- ***Scenarios 3 and 4.*** *Home composting, infrequent mixing:* Home composting with mixing of the material every sixth week.

- ***Scenarios 5 and 6.*** *Home composting, no mixing:* Home composting with no mixing of the material.
- ***Scenario 7.*** *Incineration Aarhus WtE 2003:* Incineration in the Aarhus WtE plant with a low energy recovery system and flue gas cleaning system.
- ***Scenario 8.*** *Incineration Aarhus WtE 2006:* Incineration in the Aarhus WtE plant with a high energy recovery system and updated flue gas cleaning system.
- ***Scenario 9.*** *Conventional landfill:* Landfilling in a conventional landfill site.

The LCA was performed by using EASEWASTE and the following assumptions made in the assessment: Peat substitution was modelled on a 1:1 volume basis (Boldrin et al., 2010a), as in the LCA on garden waste in Aarhus. The actual substitution of peat by compost was estimated from the user surveys at 21% (Andersen et al., VI). Fertiliser substitution was modelled based on nutrient content in the compost and the utilisation rate. The utilisation rates were assumed to be 20% for N and 100% for P and K. The actual substitution of fertilisers was estimated at 18% in the user survey on compost use (Andersen et al., VI).

Thermal treatment was performed at the same WtE plant as in the LCA on garden waste treatment in Aarhus (Aarhus 2006). Additionally, another WtE plant was modelled to show a range of Danish incinerators. The Aarhus WtE plant was used again, but before introducing the additional flue gas cleaning system and improved energy recovery (the scenario is called 'Aarhus 2003'). Efficiencies were measured at that time (2003) as 11% for electricity and 69% for heat. The two WtE technologies represent an 'efficient' and a 'less efficient' technology and can be seen as a kind of sensitivity analysis, since a range is presented. The landfill was modelled as a conventional landfill with energy recovery. The facility was equipped with a bottom liner, leachate collection system and leachate treatment, top soil cover, gas collection system, flares and gas utilisation for energy recovery (Manfredi and Christensen, 2009).

4.2.2 Results

The modelling of the home composting scenarios showed that the main processes contributing to environmental loads were GHG emissions during composting (contributing to GW) and arsenic, chromium and mercury leaching when

compost is used on soil (contributing mainly to HTs). The largest environmental savings are from fertiliser substitution (in HTs and HTw due to savings in chromium emissions from avoided P fertiliser production) and peat substitution (in GW due to saved CO_2 emissions from avoided peat use). The other potential impact categories are relatively insignificant for home composting, partly due to avoided collection and transportation of the waste. The total potential impacts are shown in Figure 14.

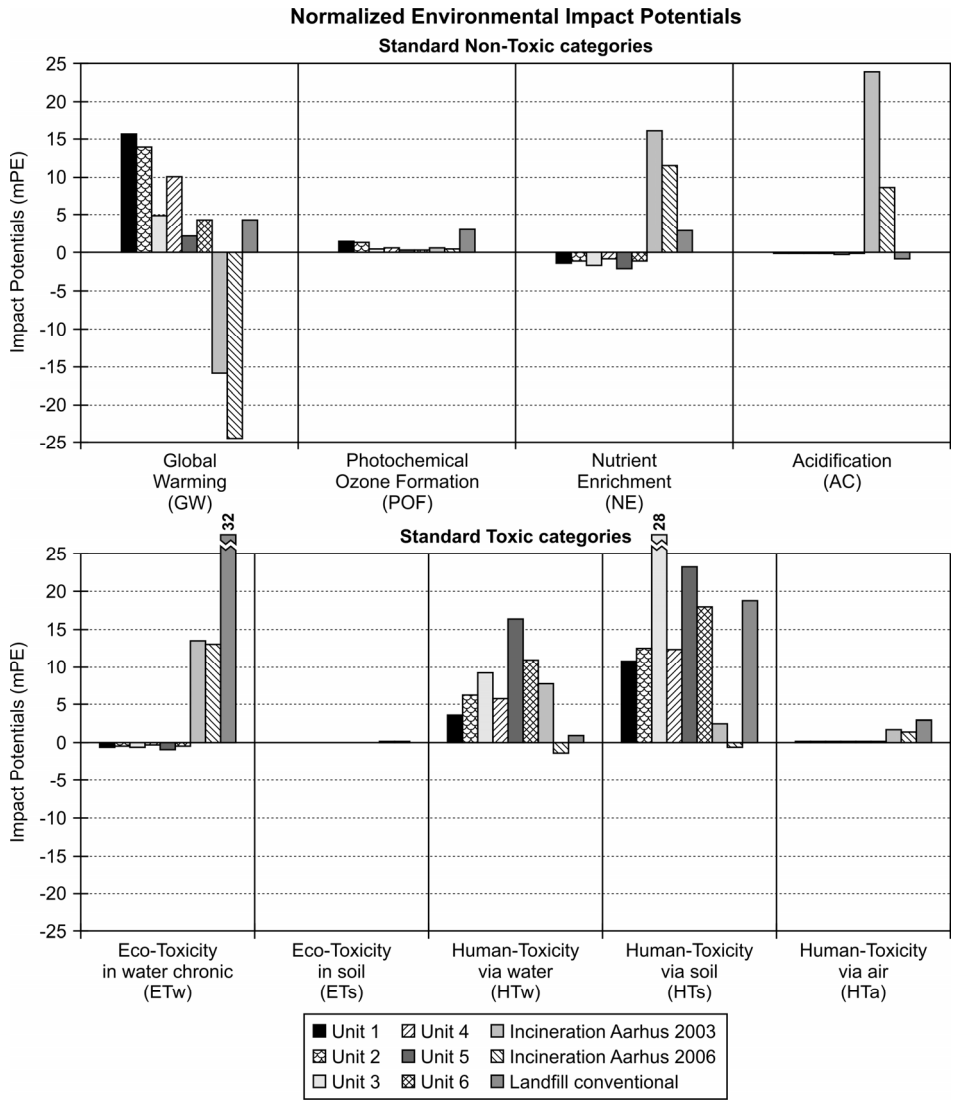

Figure 14. Comparison of potential non-toxic (top) and toxic (bottom) environmental impacts for the management of OHW by home composting, incineration and landfilling (Andersen et al., VIII).

When comparing the home composting units, it is interesting to see the large difference in GW contributions, due to the relatively large difference in CH_4

emissions from the units, caused by the variation in mixing frequencies. Units 1 and 2 (frequently mixed) had by far the highest emissions of CH_4, whereas the emissions from Units 5 and 6 were low (no mixing) (Andersen et al., III). Incineration is by far the best management option in the GW category due to electricity and heat utilisation (substitution of coal-based energy). The performance in GW by the landfill is comparable with home composting Units 3 and 6.

Contrary to GW, environmental contributions from most of the other impact categories (especially the toxic impact categories) depended on the input. For example, the most important substances contributing to HTs and HTw were arsenic, chromium and mercury, which do not degrade and therefore end up in the compost that is applied to soil. The input concentrations of these compounds thus directly influence the impacts shown in Figure 14. The lowest input concentrations of arsenic, chromium and mercury were found in Units 1, 2 and 4 (Andersen et al., IV), where the lowest impacts (in HTs and HTw) are also found. Heavy metal release when compost is applied to soil highly affects some impact categories. However, all concentrations were found to be below threshold values (Andersen et al., IV) and therefore not considered to constitute any problems when applied to soil.

Environmental loads from home composting were relatively insignificant in most impact categories. Home composting performed as good as or better than incineration and landfilling in many potential impact categories. Especially in the impact categories AC, NE and ETw, home composting was superior to incineration and landfilling (not for AC though). The main environmental loads from incineration were NO_x (NE), NO_x and SO_2 (AC) and PAH (ETw) in the stack emissions and from emissions related to diesel combustion. Incineration performed better than home composting in the potential impact categories GW, HTs and HTw.

The most critical parameters for home composting were found (from a sensitivity analysis, where the systems were modelled with changes in key parameters) to be peat and fertiliser substitution and gaseous emissions. The former was assumed to be quite uncertain but important, since the potential for increasing environmental performance is large, which indicates that the application of compost should be optimised by substituting more peat and fertilisers. Another

critical parameter was, from the sensitivity analysis, found to be the emission of NH_3 since it affects NE and AC significantly. This was shown in the sensitivity analysis by including a loss of NH_3 equivalent of 6.3% of the total N content of the input waste (based on Amlinger and Peyr, 2002). By including NH_3 emissions, the units for the impact categories AC and NE were at the same level as for incineration and landfilling. Another sensible parameter in the modelling was the choice of substitution mechanism for the WtE plant. Marginal heat production was changed to biomass (instead of coal), in order to mimic already planned future changes in the Aarhus power plant. This change decreased the environmental performance of incineration in most potential impact categories.

In general, home composting performed as good as or better than incineration and landfilling in several impact categories. It should therefore be promoted on a municipal level as a supplementary treatment of the OHW.

5 Discussion

5.1 Environmental aspects of composting

The GHG emissions, was for the first time measured with a total emission method at a full-scale composting facility (treating garden waste), and validated through a well-documented carbon balance. Other authors that have assessed GHG emissions from composting of garden waste have reported EFs within the same order of magnitude as the 111±30 kg CO_2-eq. Mg^{-1} ww reported in this thesis. Hellebrand (1998) estimated (with a gas transport model) a total EF of 194 kg CO_2-eq. Mg^{-1} ww for windrow composting of garden waste, whereas Amlinger et al. (1998) reported (from measurements in static open emission chamber) an EF of 68 kg CO_2-equivalents (eq.) Mg^{-1} ww for windrow composting of garden waste. These EFs, as well as all other GW contributions were assessed in detail in the thesis, since they are important in environmental assessments and have not been assessed in detail and validated from full-scale composting operations. However, a more holistic view of environmental performance should be taken, since GW does not always represent the overall performance of waste management systems. Merrild (2009) showed that the representativeness of GW as an indicator for the environmental performance of waste management systems was quite poor, and suggested that GW should not stand alone. This is in line with findings in this thesis, since incineration was better than composting when looking only at GW, whereas composting was better in several of the other categories. An even more holistic approach would be to include economic and social assessments of waste management.

In general, the environmental performance of composting (both central and home composting) was found, from the LCA modelling, to be quite good. The potential environmental impacts of composting were found to be relatively small - in the range -6 to 8 mPE Mg^{-1} ww for non-toxic categories and 4 to 100 mPE Mg^{-1} ww for toxic categories for central composting of garden waste and -2 to 16 mPE Mg^{-1} ww for non-toxic categories and -0.9 to 28 mPE Mg^{-1} ww for toxic categories for home composting of OHW. The potential non-toxic impacts for central and home composting were smaller than found for treatment of other types of MSW (Kirkeby et al., 2006a), whereas the potential toxic impacts were in the same level, especially for central composting of garden waste. This was due primarily to an increased content of heavy metals (especially chromium, arsenic and mercury) in the garden waste compost compared to OHW compost (as well as

toxic effects from combustion of fossil fuel). The reason for the higher impacts for the toxic categories is also related to issues with the LCA method when calculating toxic impacts, which is discussed in more detail in Section 5.2.

One outcome of the two LCAs was that composting is environmentally comparable to or better than alternative treatment options for organic waste in several potential impact categories. The most significant contributions to environmental impacts were loads from direct GHG emissions produced by the composting process itself (direct impacts) and loads and savings in emissions from applying the compost on soil (indirect downstream impacts). These two processes should be optimised in order to improve the environmental profile of composting. There is good potential for decreasing the emissions from the facility, whereas it is more difficult to optimise the use of compost. Home composting should be seen as a good supplementary treatment for part of the organic waste stream (especially in rural areas) and the LCA of home composting of OHW showed no indications that the impacts were larger than for central composting.

Martínez-Blanco et al. (2010) considered several non-toxic categories in an LCA on home composting and industrial OHW composting (with biofilters), the impacts of which were in the same order of magnitude as found in this thesis (for GW, AC and NE which were the only mutual impact categories). Indirect upstream processes like the home composting units and tools used during composting were included in the study by Martínez-Blanco et al. (2010), but they constituted less than 10% of the total impacts in the categories GW, AC and NE. No indirect downstream processes were included in that particular assessment.

Changes in the Biomass Ordinance in Denmark (as mentioned in Section 1.2) made incineration of garden waste interesting for waste managers. It is, however very important to clarify that it is not recommended to incinerate all garden waste, since the ash content is very high (48.3% on average during the year as stated by Andersen et al. II) and the LHV quite low (6.4 GJ Mg^{-1} ww on average during the year as stated by Andersen et al. II). This means that garden waste should be sorted out, for example, in the winter months where the proportion of large woody parts is highest and screened to obtain a suitable refuse derived fuel (RDF) fraction for incineration. Initial incineration tests (with RDF from garden waste) were performed by Vestforbrænding (VF) I/S (a publicly owned waste

management company in the Copenhagen area, Denmark), which showed that un-shredded and unscreened garden waste was not suitable for incineration in WtE plants. There were on the other hand, no problems with pre-treated (shredding and screening) garden waste in the incineration test, but the LHV was still low (around 10 GJ Mg^{-1} ww) (personal communication with Alan Sørensen, Vestforbrænding, 2010), and therefore should not be seen as a high priority waste fraction for incineration but instead as a supplement (e.g. when normal MSW production is low). If all garden waste was incinerated (in addition to problems in the furnace), all nutrients would be lost and there would be no compost available for use on soil. One of the outcomes of the user surveys was that the compost users were quite content with the service provided by the municipality (pick up of compost at the RC), and it was indicated by some of the people in the home visits that the compost facilitated aesthetic improvements when applied in the garden, such as nicer soil colour and better plant growth. Thus, it is still important to maintain the composting process.

The choice between different options for the treatment of organic waste (AD, composting, WtE energy, etc.) should be taken based on local conditions (climate, energy mix, local policies) since this will vary greatly, even within the borders of the EU. Waste incineration for example is better suited for countries in colder climates, where heat production can be utilised effectively (European Commission, 2010). To illustrate this point, there is a long tradition in Denmark for waste incineration, and WtE technologies are efficient with very high energy recovery rates. This is the reason why incineration performs as well as composting (in some of the impact categories) in the LCAs presented in this thesis. This probably would not be the case if the comparison was made in countries with less efficient incinerators or less need of heat production.

5.2 Data uncertainty

The data collected for central and home composting was in many ways more comprehensive than what has previously been reported in the literature. Some of the data is thought to be relatively certain and some data quite uncertain. The direct emission data (consumption of fuel and electricity) provided by the composting plant and the indirect upstream emission data (provision of fuels and electricity) are easy to obtain and the uncertainty is small. The GHG emissions were assessed in detail and the EFs generated were believed to be good estimates of the actual emissions. For full-scale windrow composting it was concluded that

on-site total emission measurement methods should be used to measure GHG emissions. Neither concentration measurements nor small-scale flux type measurement methods were sufficient to estimate the actual gaseous emissions. EFs reported for the composting plant in Aarhus from the dynamic plume method were slightly overestimated but considered the most robust method. There are quite a few uncertainties related to these kind of measurements such as determining the actual amount of waste present at the site (when relating the emissions to the total amount of waste). The dynamic nature of the emissions from composting windrows is also affected by meteorological conditions such as barometric pressure and wind speed. Finally, measurements should be carried out many times during the year to represent an average annual release of gases. Thus, the EFs were considered satisfactory. The EFs for home composting were, on the other hand, easier to measure and the dataset far more robust (representing the temporal variations well) than the data for central composting. The experiments with higher input loads into the home composters indicated that emissions increase with increased input. This was, however, not seen from the high-input home composting experiments by Amlinger et al. (2008) and Martínez-Blanco et al. (2010), who presented EFs in the same range as in this thesis. NH_3 emissions were found to be of limited importance in relation to central composting of garden waste and home composting of OHW. However, sensitivity analysis showed that NH_3 emissions could potentially influence environmental performance significantly, and should thus be mapped in more detail from both central and home composting perspectives.

Conversely, environmental savings from compost use are more uncertain and greatly determine the environmental profile of composting. Estimated substitution rates of peat, fertiliser and manure were obtained from user surveys, but should be regarded as first estimates, as far more data collection is needed to decrease their uncertainty. The substitution mechanisms were assumed to be the same for compost produced from garden waste (central composting) and OHW (home composting). Knowledge about compost produced from home composting is, however, even more limited than from central composting. Thus, the estimated substitution of peat, fertiliser and manure might be overestimated for home composting. The quality of the compost produced at the central composting plant was also considerably better than that produced at home, which means that garden owners probably would be keener on using compost produced

centrally. This is, nonetheless, speculation, and more investigations would have to be carried out to support this idea.

Potential toxic impacts were quite high (especially for HTs) in the assessment of garden waste management in Aarhus, as the compounds dominating the toxic categories were heavy metals (chromium, arsenic and mercury). The content of heavy metals was, in general, higher in garden waste compost compared to OHW compost, which resulted in higher potential impacts from garden waste compost. This makes sense, since the ash content (sand and soil) in garden waste compost is very high and heavy metals are bound to the soil particles. The results for the potential toxic impacts, though, seem to be overestimated, since the LCA methodology for compost use estimates potential toxic effects based on amounts (instead of accounting only for effective concentrations) of heavy metals applied on to soil. This means that the full amount of each heavy metal is accounted for in the toxic categories, even though they are below the threshold values. The compost produced in Aarhus respects all quality standards (Andersen et al., II) for heavy metals and thus does not constitute a problem when applied to soil. The LCA methodology should be optimised to better represent the spread of heavy metals in soil.

5.3 Missing aspects

Many potential benefits from composting were not included in the environmental assessments due to difficulties in quantification, but all are linked to the application of compost on soil:

- Reduced water use (from better water holding capacity of soil)
- Reduced fertiliser, herbicide, pesticide, lime and gypsum requirements
- Improved soil properties and thus better plant growth

These benefits (and others) were included in an LCA on windrow composting by ROU (2006). Saved amounts of water and materials/products were reported, but not included in the potential impact categories. These benefits are most likely of minor importance when added to all the other contributions from the actual composting process, but they should still be included in order to contribute to a full environmental assessment of composting. Methodologies that include these benefits should be developed in an LCA context.

The benefits that can be obtained by applying compost on soil very much depend on where it is applied. For example, no benefit can be obtained from the use of garden waste compost in agriculture, since it is not a part of the fertiliser schemes (Plantedirektoratet, 2006). This means that, garden waste compost that farmers apply on their farm-land is a supplement to the fertilisers they would use anyway. Compost use on farm-land could result in additional plant growth, but this is relatively difficult to quantify. Thus, the only benefit from using garden waste compost in agriculture is carbon binding. Benefits from substituting peat can be achieved if the compost is used instead in hobby gardening or horticulture.

In addition to the increased focus of using garden waste as an RDF for producing electricity and heat in WtE plants, other treatment options are being considered at the moment in Denmark. Direct application of shredded garden waste on agricultural fields has been tested and the first environmental assessment recently published (Møller et al., 2010). The results indicate that the environmental performance of direct application is more or less equal to windrow composting followed by application on agricultural soil. It is noteworthy that it is only possible to apply the garden waste directly on to soil for short periods of the year according to agricultural practices, which means that it should be seen as a supplementary technology to central composting. These recent technologies (incineration and direct application) for garden waste treatment have not been fully assessed.

Since both nutrient recycling and energy recovery are high on the political agenda, more research emphasis should be placed on combined technologies for the treatment of organic waste. In combined technologies, energy is utilised from AD of the waste, followed by composting of the residues to produce a nutrient-rich end product that can be used on soil.

5.4 Recommendations

It is recommended that the size of the windrows at Aarhus composting plant be decreased to a maximum of 2.5 m. This will enhance air intrusion in to the compost material and thus lower the emissions of CH_4, consequently making turning easier (e.g. with an automatic windrow turner instead of the current method of turning with a front loader). This is a qualified guess, though, since no measurements were performed at smaller windrows. However, experiments by Beck-Friis et al. (2000) indicated that concentrations of CH_4 increase with

increasing windrow height. In the case of restructuring the windrows, management would change, since the decrease in dimensions would change the properties of the waste, e.g. moisture content and temperature would be different and this would influence the whole composting process. The period of composting in Aarhus is very long, and so it is recommended that the total composting time be lowered, which means that less space would be needed or more waste processed at the facility. It is also recommended to use mature compost more rationally, i.e. to use the compost so it actually substitutes for peat or fertilisers. It should still be possible for the citizens of Aarhus (and everywhere else) to pick up compost at their local RC, but a part of the produced compost could be used professionally, in order to reap the environmental benefits of peat and fertiliser substitution - it is very difficult to 'teach' every citizen the importance of peat substitution, while it would be far easier to convince professional gardeners.

Home composting should be promoted on a municipal level, and it is especially important to promote home composting in remote regions (low density areas) where the collection of waste results in consuming a large amount of fossil fuel due to transportation distances between each pick up point. Home composting take place in many different management schemes, but in general it does not seem to be an environmentally problematic process, except for the very eager mixing of the material in the composting units. It is recommended to mix the material just once per month or so and definitely not every week (even though the decomposition of the organic matter will be slower).

6 Conclusion

Composting of organic waste entails GHG emissions, which cannot be avoided entirely, even under well-aerated conditions. CH_4 and N_2O especially are generated in significant concentrations and are at least partly emitted into the atmosphere, thereby contributing to GW. Contributions to GW from windrow composting of garden waste were assessed comprehensively in a field study in Aarhus, Denmark. Two small-scale (static flux chamber and funnel methods) methods and one total emission (dynamic plume method) method were applied to quantify GHG emissions from the facility. The small-scale methods clearly underestimated the emissions and were found to be inappropriate for quantifying GHG from windrow composting of garden waste.

The emission estimate was instead made by means of the dynamic plume method, where a tracer gas release is combined with concentration measurements downwind of the composting facility. The total EF was quantified as 111 ± 30 kg CO_2-eq. Mg^{-1} ww (based on a release of 2.4 ± 0.5 kg CH_4-C Mg^{-1} ww and 0.06 ± 0.03 kg N_2O-N Mg^{-1} ww). This was the first time that the GHG emissions were measured from a full-scale composting plant with a total-emission measurement method. It was concluded, from testing a gas transport model, that it is not enough to do concentration measurements in the compost pore space if we need to make realistic estimates of the emissions; on-site emission measurements are needed.

The EFs from home composting of OHW were quantified as 100-239 kg CO_2-eq. Mg^{-1} ww from six differently managed home composting units. The variation in EFs from these was due mainly to differences in the mixing frequencies of the material in the composting units. The highest mixing frequencies entailed the highest emissions of CH_4. The composting unit that was managed in the most realistic way (infrequent mixing and low addition of waste) had the lowest GHG emissions (127 kg CO_2-eq. Mg^{-1} ww) and the GHG emissions from home composting were comparable with those from central composting (open technologies) of garden waste. The feedstock was not the same in the two cases and the GHG emissions could potentially be higher for central composting of OHW (compared to central composting of garden waste and home composting of OHW). Aeration/mixing frequency seems to be of great importance in relation to gaseous losses when composting organic waste. In addition, volume, surface area

and compaction of the waste have great importance for the generation and emission of GHG. The GW EF that was provided in this thesis can act as robust input parameters for environmental assessment of composting technologies.

In addition to direct GHG emissions during the degradation of organic waste, other processes related to the composting process contributed indirectly to GW. The most important process was identified as the substitution of fertilisers and peat in growth media when compost is used in private gardens (or in horticulture). The potential benefits are great - up to 880 kg CO_2-eq. Mg^{-1} ww - assuming that all compost produced is used as a substitute for peat. This means that there is potential for composting to be a total environmental saving (instead of an environmental impact); however, the actual use of compost as a substitute for fertiliser and peat is not a well researched topic. In this thesis, it was estimated by performing user surveys that the substitution of fertiliser and peat, when using compost in private gardens, was 18 and 21%, respectively. This is still far from the potential and there is thus a great opportunity to impact total GW contributions by optimising the use of compost.

In order to take a holistic environmental view of organic waste composting, and thereby look at potential impact categories other than GW, LCAs were carried out, thereby assessing the full environmental performance of composting. Firstly, full LCIs were generated for windrow composting of garden waste and for home composting of OHW. In both cases, all direct emissions were reported and MFA/SFA was used to follow the flows of nutrients (N, P and K) and heavy metals (mainly Cd, Cr, Cu, Hg, Pb and Zn) especially. LCIs were used as a platform for LCA modelling. In general, it was found from the LCA modelling of garden waste treatment in the LCA-waste-tool EASEWASTE that for most impact categories it was beneficial to take out woody material and incinerate it in a WtE plants. It is, however, questionable whether it would make sense to incinerate all garden waste, due to the relatively low LHV and the high content of ash. For the LCA of home composting of OHW, incineration performed better than home composting in the GW impact category, whereas home composting performed as good or even better as incineration in many of the remaining impact categories.

There is still a range of factors that are not accounted in environmental assessments of composting. Some of these factors are the garden owners'

satisfaction with the service provided by the RCs and aesthetic benefits such as nicer soil colour and more beautiful flowers when applying compost in gardens.

The use of compost should be optimised in order to increase the environmental performance of composting. One important point, when using garden waste compost in agriculture, is that farmers do not have to include it in their fertiliser plan. This means that they use garden waste in addition to normal fertilisers, and therefore no credits are obtained for the substitution of fertilisers. In the case of the agricultural use of garden waste compost, the only credit obtained is from carbon storage in the soil. However, compost application could result in an increase in crop production; in some cases it would make better sense to use garden waste compost in private gardens or in horticulture as a substitute for growth media (including peat) and fertilisers.

In general, small environmental impacts are created by composting (central and home composting) of organic waste (garden waste and OHW). Potentially there are great benefits if compost is applied instead of commercial products such as peat and fertilisers. The results from the environmental assessments indicated that home composting should be promoted, since environmental performance is comparable to the alternatives and, in some cases, even better.

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8 Appendices

I Andersen, J.K., Boldrin, A., Samuelsson, J., Christensen, T.H., Scheutz, C., 2010. Quantification of GHG emissions from windrow composting of garden waste. Journal of Environmental Quality. 39:713-724.

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The papers are not included in this www-version, but can be obtained from the Library at DTU Environment. Contact library@env.dtu.dk or Department of Environmental Engineering, Technical University of Denmark, Miljoevej, Building 113, DK-2000 Kgs. Lyngby, Denmark.

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